BC Environment

Timber-harvesting Effects on Riparian Wildlife and Vegetation in the Okanagan Highlands of British Columbia

by
Les W. Gyug

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British Columbia, Canada's most westerly province, has a bounty of biological diversity. British Columbia's snowclad peaks, rain-drenched forests, arid grasslands, all sizes of rivers, lakes, and wetlands, and a long and rugged coast provide habitats for more species of living organisms than are found anywhere else in Canada. However, this very diversity means that there is much to be discovered about these organisms — their distribution, abundance, habitat requirements, and interrelationships with their environment. Increasing our knowledge of this biodiversity will help us with the complex task of sustainably managing our land and waters.

In 1992, the Provincial Government initiated a co-operative biodiversity research program with funding from the Corporate Resource Inventory Initiative, the British Columbia Ministries of Forests (Research Branch), Environment, Lands, and Parks (Wildlife and Habitat Protection Branches), and Tourism and Culture (Royal B.C. Museum), and the Forest Resource Development Agreement (FRDA II).

In 1995, the Ministry of Forests Research Branch and the Ministry of Environment, Lands and Parks developed a biodiversity research and extension strategy, with the assistance of the provincial research community. This strategy was presented to Forest Renewal BC (FRBC), who provided funding for a program beginning in 1995. The goal of the extension component of this program is to extend information to scientists, resource managers, and the public through biodiversity publications. These publications are intended to increase awareness and understanding of biodiversity, promote the concepts and importance of conserving biodiversity, and communicate provincial government initiatives related to biodiversity. We hope that they will be used as tools for the conservation of British Columbia's rich, living legacy.

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TIMBER-HARVESTING EFFECTS ON RIPARIAN WILDLIFE AND VEGETATION IN THE OKANAGAN HIGHLANDS OF BRITISH COLUMBIA

by
Les W. Gyug

BRITISH COLUMBIA

Ministry of Environment, Lands and Parks
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FOREWORD

The views expressed herein are those of the author and do not necessarily represent those of the B.C. Ministry of Environment, Lands and Parks.

In cases where a Wildlife Bulletin is also a species' status report, it may contain a recommended status for the species by the author. This recommendation is the opinion of the author and may not necessarily reflect that of the Wildlife Branch. Official designation will be made by the Wildlife Branch in consultation with experts, and the data contained in the status report will be considered during the evaluation process.

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EXECUTIVE SUMMARY

Riparian areas often support the highest diversity and abundance of wildlife in temperate zones because of their diversity of natural features, because they can support species uniquely adapted to the riparian habitats, and because they can support upland species that may depend on riparian habitats for part of their life cycle. Riparian areas are vital parts of an ecosystem acting as connecting areas for surrounding uplands for water and nutrient flows. Clean, clear water is also essential for healthy fish populations. For these reasons and more, riparian areas are often singled out as needing special protection and management. Rautio and Bunnell (1992) identified the greatest single problem in managing riparian ecosystems in B.C. as the lack of basic information on size, shape, species composition, and ecosystem responses. They stressed the need for baseline research on wildlife habitat, biodiversity and ecosystem functions, and on the impacts of forest management regimes.

The purpose of this overall research project was to identify species for which riparian ecosystems were relatively important, and to provide recommendations for forest management that will best protect these species.

There has been very little research in B.C. on the influence of timber harvesting on the diverse wildlife of riparian areas (Rautio and Bunnell 1992). The objectives of the research phase of this project were first, to describe the difference in riparian wildlife and vegetation of the Okanagan Highlands compared to uplands; second, to describe the effects of clearcutting on riparian and upland wildlife and vegetation; and third, to describe the effects of retention of forest corridors of various widths and levels of partial harvest on riparian wildlife and vegetation. The research examined each of the three objectives for breeding birds and small mammals, and a more limited set of objectives for the natural cycles of trees and coarse woody debris (CWD), for amphibian breeding in forests and clearcuts, and for winter-active wildlife use of riparian zones under different timber harvest treatments. Results for each of these major study groups are reported here in chapters.

The objective of the second phase was to develop timber-harvesting guidelines that would maintain the natural diversity of riparian areas in the Okanagan Highlands, and to compare these guidelines to the riparian timber-harvesting guidelines of the Forest Practices Code of 1995.

STUDY AREA

The study area was within the Montane Spruce (MSdm1) and Engelmann Spruce–Subalpine Fir (ESSFxc and ESSFdc1) biogeoclimatic zones in the Okanagan Highlands of southern B.C. The study area is a high plateau (1300-2000 m) with numerous small lakes, wetlands, and low-gradient streams. Sampling sites of 1 to 5 ha in size, depending on taxon studied, were used from throughout the area. Internal validation of the results (section 7 of this report) through intensive sampling of two small, second-order sub-drainages (Echo Creek 917 ha, Alex Creek 1263 ha) was also undertaken to check the predictive power of the results within the study area. Note that in this executive summary, only those instances where results differed significantly from the main study are included.

VEGETATION

Trees and Coarse Woody Debris (CWD)

Riparian forest vegetation of the MSdm1 is dominated by spruce (Engelmann-white hybrid) or subalpine fir trees. These stands usually exist in sharp contrast to the adjacent upland stands where lodgepole pine predominates in a fire-caused disclimax forest. This pattern is typical of lodgepole-pine-dominated regions of the western cordillera. The objectives of this phase of the study were:

- To quantify the riparian-upland tree species difference and its effect on natural cycles of wildlife trees (dead standing trees >10 cm dbh) and CWD (downed logs >7.5 cm diameter).
- To determine to what extent the riparian-upland species difference was caused by trees surviving fires or by preferential establishment of spruce in riparian sites after stand-replacing fires.
- To contrast this to the ESSFxc Zone where spruce and subalpine fir were more common in upland forests than in the MSdm1.
- To delineate natural successional and disturbance patterns that might serve as models for riparian timber-harvesting guidelines.

Seventeen 1-ha (40 × 250 m), randomly selected, streamside plots in the MSdm1, and six in the ESSFxc, were paired with adjacent 1-ha upland plots. Veteran trees and large wildlife trees (>30 cm dbh) were counted in the 1-ha plots. Canopy and sub-canopy trees were counted in four 0.16-ha (10 × 40 m) subplots within each 1-ha plot. CWD volumes were estimated on a 90-m transect parallel to the stream within each plot.

The tree canopies of young and mature (54-192-year-old) streamside MSdm1 forests were less dense than upland forests (750 vs. 1261 trees/ha), and were a mix of species dominated by spruce (48%), followed by lodgepole pine (39%), and subalpine fir (12%). In contrast, the upland forests were dominated by lodgepole pine (92%), with spruce being very uncommon (6%). In 20% of the riparian sites, topographic effects on fire behaviour were the sole source of this diversity. Gullies deeper than 28 m and floodplains wider than 50 m appeared to deflect or reduce the intensity of fires, leaving older stands in these riparian sites. Tree species diversity in the remaining 80% of the riparian stands resulted mostly from the establishment and growth of trees in the post-fire stand.

The best descriptor of floodplain forest was the hygric/subhydric site series — in this case site series 07 of the MSdm1. These floodplain forests were characterized by trees growing on organic hummocks, or on sandy or silty alluvial deposits with very high water tables <30 cm from the surface. Stream width was not a good predictor of the extent of riparian floodplain forest (r^2=0.00) in the first- and second-order streams studied.

Old-growth upland stands in the MSdm1 were defined as those older than 250 years. Only after 250 years did the upland stand structure and tree composition differ markedly as spruce and subalpine fir replaced senescent lodgepole pine. Young and mature streamside stands often had the appearance of old-growth stands because spruce was dominant, trees were larger, canopy trees were less dense, and there were more sub-canopy trees and understorey trees. Spruce, in particular, grew faster in streamside stands, averaging 5.0 cm dbh larger than upland-grown spruce at ages of 100-120 years. Canopy tree density was consistently low in old-growth stands (374/ha) and more similar to the younger streamside stands than the younger lodgepole pine upland stands. There were fewer sub-canopy trees in the young upland stands than in the old-growth and the young streamside stands (both in absolute terms and in proportion to the densities of canopy trees). Because of these similarities, wildlife that require spruce-fir forests may find young and mature riparian MSdm1 forests as suitable as old-growth forests.

The riparian spruce-upland pine pattern did not necessarily continue upward in elevation into the ESSFxc. In the main study, young upland ESSFxc forests were as diverse in structure and tree species as the riparian forests, but in the two selected sub-drainages, the MSdm1 pattern did continue into the ESSFxc. Local effects and local fire history may also influence the riparian-upland effects on tree distribution in the ESSF.

Natural cycles of large wildlife tree densities and CWD volumes were highly correlated with stand age. Immediately after a fire, levels of large wildlife trees will peak in the forest of standing grey trees remaining after a fire. As these trees fall over, CWD will peak at levels similar to old-growth forests. Both of these cycles will be at low points around 100 years after a fire, as most wildlife trees will have fallen over and most CWD will have decomposed. Both of these levels then increase as the forest matures, and the trees that established and grew after the fire start to self-thin or die.

Wildlife tree and CWD levels in any young stand also depended on the size of the trees in the pre-fire stand. The larger the trees in the pre-fire stand, the slower the decline in wildlife trees because the larger trees would fall over more slowly, and the CWD decay would be spread over a longer period. The legacy of pre-fire riparian stands >150 years old continued for over 100 years into the post-fire stand. The same effect in upland stands was only noticed when the pre-fire stand contained large Douglas-fir, or was old growth. Large wildlife trees were more abundant in riparian stands than in upland stands. Mean log diameter was higher in riparian stands, but CWD volumes were not significantly higher.

Clearcut logging is projected to set wildlife tree and CWD levels into a permanently very low cycle that naturally occurs in very few cases (i.e., where two fires may have naturally occurred on the same site within a period of less than 50 years). Any partial harvesting will result in a reduction in wildlife trees as they are felled to meet Workers' Compensation Board regulations. The degree of the reduction will depend on how much of the area is declared a work zone. The effects of partial harvesting on CWD levels are more difficult to predict, but should deviate less from the natural cycle.

Wildlife tree use differed by size and tree species. Four percent of large wildlife trees (>30 cm dbh, N=602) contained nest holes, while only 0.08% of small wildlife trees (10-29 cm dbh, N=1311) contained nest holes. Woodpeckers foraged on twice as many large wildlife trees as small wildlife trees. The most highly used was large Douglas-fir (88% used), which was rare in the study area, accounting for only 2% of the wildlife trees and 1% of the live trees tallied. Next were large spruce (69%), large lodgepole pine (69%), small spruce (42%), small lodgepole pine (27%), large subalpine fir (24%), and small subalpine fir (10%), in decreasing order of use.

Understorey vegetation

Vegetative diversity was assessed in the MSdm1 at three sites each in clearcuts, partial cuts, and uncut controls. At each site, paired samples within the creek floodplain area (riparian habitat) and in the adjacent area above the first prominent terrace (upland habitat) consisted of four 400-m^2 (40 × 10 m) plots for shrubs and four 4-m^2 (2 × 2 m) subplots for herbaceous species. Ground cover (%) of all species was estimated visually.

A total of 100 vascular species were recorded: 66 occurred on both floodplain and upland sites, while 19 were restricted to floodplains and 15 to upland areas. Riparian sites consistently had more species (39% vs. 30), higher herb cover (72% vs. 50%), and higher shrub cover (26% vs. 11%), whether the sites had been clearcut (with understorey reserve), partially cut, or totally reserved. Moss cover was significantly higher in uncut upland forests than in uncut riparian forests (55% vs. 10%). In riparian sites, shrubs and herbs rather than moss made up the majority of ground cover. Clearcutting or partial cutting of upland habitats eliminated much of this moss cover with a resultant increase in herbaceous cover. Clearcutting or partial cutting of riparian habitats tended to result in increased shrub cover.

BREEDING BIRDS

Breeding bird densities in riparian and upland areas were compared in clearcuts, forest corridors, and unharvested forests. The effects of corridor width and partial harvesting in corridors on bird densities were also assessed. This study covered the MSdm1, the ESSFxc, and the ESSFdc1.

Breeding birds were sampled during May and June of 1994 and 1995 using all bird detections from six visits to each of the eighty-four 5-ha (500 × 100 m) plots. Individual plots were only sampled in one of the two years.

Riparian areas were very important to the bird communities of the Okanagan Highlands, where 37% of the 101 bird species encountered were classified as riparian-dependent. Six of these species depended on riparian forest in the MSdm1. For these species, and a few other forest species, the riparian MSdm1 forests were essentially downward extensions of more suitable ESSF forest habitat where spruce and subalpine fir were abundant. Eight species required trees for nesting or perching near riparian areas, and the American Dipper required healthy streams for foraging on invertebrates. The other 22 riparian species were wetland or lake species that did not appear to be negatively affected by timber harvest in adjacent areas. Swifts and swallows made up another 7% of the species total and, while not always riparian-dependent, they may spend considerable foraging time in wetlands or along streams. Consequently, riparian areas have an importance to birds that far outweighs their actual extent: up to 44% of the birds rely on habitat that occupies only 5-10% of the area.

No species could be identified as upland-dependent. Of the remaining 56% of the species, only the Red Crossbill was significantly more abundant in upland forests than riparian forests. All these species appeared to use both riparian and upland habitats.

Using unharvested forests with a minimum 500-m forest width as controls, edge-effects in species composition were identified and abundance in the widest category (132 to 280 m width) of riparian forest corridors was examined. Densities of edge species (able to use very open forests or clearcuts) increased from 21% to 48% of total densities. A threshold effect was found in riparian forest corridors at 70 m, where edge species densities increased to 60% and five forest bird species declined in density — Winter Wren and Townsend's Warbler (which relied on riparian forests in the MSdm1), and the Boreal Chickadee, Brown Creeper, and Gray Jay (which were also common in upland forests, so that riparian forests would not be essential in their management). Another threshold occurred at 38 m where riparian forest corridors completely lost suitability for the 5 species listed above and for 10 others typically limited to closed-canopy forests. Upland forest corridors were far less diverse than riparian forest corridors, and were dominated by Dark-eyed Juncos.

Partial harvesting at low levels (<33% stem removal) appeared to have very little effect on bird communities in wide streamside forest corridors (69 to 150 m). However, high levels (48-85%) of removal in narrow corridors (47-55 m) had significant effects and resulted in the absence of the six forest species most sensitive to corridor width at the 72-m threshold, and 70% declines in densities of other forest bird species.

Wildlife trees were an essential habitat component for the bird community of the Okanagan Highlands. Twenty-two percent of the 100 breeding species encountered and 46% of the permanent resident (i.e., wintering) bird species were cavity-nesting birds. No sampling was done for owls, many of which are cavity nesters.

SMALL MAMMALS

Objectives were to determine the species richness, composition, and abundance (1) in upland and riparian areas in clearcuts and forest corridors compared to uncut controls, and (2) in forest corridors of various widths and levels of partial harvest. Small mammals were sampled on snap-trap transects, consisting of 25 pairs of traps set at 10-m intervals for two nights, set parallel to streams or in upland habitats. Each transect was a 100 "trap-night" sample.

The Water Shrew was the only species limited to riparian areas, but although not found in clear-cuts, the effects of clearcutting remain uncertain because Water Shrew were very uncommon in any habitat. Three small mammal species (Meadow Vole, Long-tailed Vole, and Vagrant Shrew) were significantly more abundant in riparian areas than upland areas. No small mammal species was limited to upland areas, and only Deer Mice were more abundant in upland areas than riparian areas.

The small mammal communities of unharvested forests were dominated by Red-backed Voles (66% of forest captures). Clearcutting turned riparian areas into Meadow Vole-dominated communities (56% of riparian clearcut captures), and upland areas into Deer Mouse-dominated communities (35% of upland clearcut captures). Total small mammal abundance was not reduced in riparian areas by clearcutting (17.8 vs. 13.2 captures per 100 trap-nights), but was reduced by half in upland areas (13.2 vs. 6.2 captures per 100 trap-nights).

Forest corridors along streams are probably one of the best options for conserving small mammal diversity, because these corridors supported all of the mouse, vole, and shrew species found in the Okanagan Highlands, whereas upland forest corridors did not. Upland forest corridors had lower diversity and abundance of small mammals than riparian forest corridors (20 compared to 4 small mammals, and 5.4 compared to 2.0 species caught per 100 trap-nights). Edge effects in species composition started to be shown in corridors as wide as 300 m. The most common forest species (Red-backed Vole) was present in all forest corridors, but was most abundant in corridors wider than 100 m.

Low partial harvesting levels (<33% removal) in forest corridors did not appear to affect small mammal communities. Higher levels of removal may have affected small mammal communities, but these effects could not be definitively separated from possible width effects in this study because of a lack of replicates.

AMPHIBIANS

Specific objectives were to determine which amphibian species were present and which species would breed in wetlands surrounded by clearcuts. Only presence or absence of larvae was measured because adults disperse widely after breeding and are not easily found. In the MSdm1, 30 wetlands and ponds in clearcuts and 15 in forests were searched for aquatic amphibian larvae in 1993, 1994, and 1995. In the ESSFdc1 and ESSFxc, 30 wetlands and ponds in clearcuts and 26 in forests were searched for larvae in 1995.

Larvae of four species of amphibians were found in both clearcut and forest breeding ponds: Western Toad (*Bufo boreas*), Spotted Frog (*Rana pretiosa*), Pacific Treefrog (*Hyla regilla*), and Long-toed Salamander (*Ambystoma macrodactylum*). All four species were widespread in the study area and occurred in a wide variety of both forested and clearcut wetlands and ponds. There was no significant correlation between distance to nearest forest or time since clearcutting and the proportion of breeding sites used by any species.

The immediate effect of clearcutting appeared to be the enhancement of the suitability of small pools (<0.02 ha and at least 20 cm deep) for egg-laying. These very small pools would have been too cold (<6°C) and therefore unusable under coniferous cover during the April to May egg-laying period. However, these very small pools in clearcuts were much more prone to drying out, and probably suffered higher predation rates on amphibian larvae than larger, more permanent wetland pools. Where these small pools were beside larger wetlands, they usually originated in industrial activity associated with road-building and clearcutting (e.g., ditchlines and machine-track ruts). These sites may have been reproductive sinks — they appeared to be more attractive to adults for egg-laying, but less suited to larval survival than the larger, more permanent pools that the amphibians would have bred in before the small pools were created.

The ultimate effects of clearcutting on amphibian population persistence are still not certain because larval survival rates have not been fully determined, the distance adults may be migrating to these breeding ponds is still unknown, and the effects of clearcutting on adult survival and on habitat suitability are unknown.

WINTER WILDLIFE

Tracks of animals made since the previous snowfall were tallied along transects set at right angles to streams and marshes in the MSdm1. Track densities (tracks/km-day) were compared in riparian buffer zones with total forest reserve, partial timber harvest, and clearcutting with understorey reserve. All birds detected within 50 m of the transects were also tallied. Transects were visited five times in the winter (December-February) of 1992-93.

No mammals with distinct dependence on riparian areas were encountered in this tracking, (i.e., mink, beaver, or otter), so that we do not have data to apply to the management of these species. However, four other mammal species were commonly encountered on winter track transects. Weasels were active in all areas, with highest use along streams and marshes that were clearcut with a no-machine buffer zone that preserved understorey along the stream or marsh edge. Small mammal sampling later in the year showed that the weasels were probably preying on the abundant Meadow Voles in the clearcut riparian areas.

Snowshoe Hares were most common in total forest reserves along riparian zones, but were found in all treatment types. Red Squirrels were absent from clearcut riparian sites, but were found in both total reserve and partially harvested sites. Red Squirrel track indices declined to zero at forest buffer widths of 12 m or less. However, upland forests also appeared to be useful to both these species, and clearcuts appeared to regain habitat suitability as little as 14 years after clearcutting, so that riparian forests did not appear to be essential in the management of these two species.

Marten used both partially cut and totally reserved forest corridors as foraging habitat, but only used one of the three clearcut riparian sites as a travel corridor, i.e., for straight-line travel where no other forest connections existed between suitable habitats. Marten did not make use of an apparently suitable riparian forest corridor site 500 m from continuous forest, but did use that corridor within 200 m of continuous forest. Landscape-level effects, as well as stand-level effects, were probably influencing Marten habitat use.

Upland forest track measurements were not made, except in uplands immediately adjacent to riparian areas; based on the literature and the habitat structure, habitat suitability of upland lodgepole pine forests in the area was probably low for marten compared to the higher suitability of riparian spruce forests.

Birds were more active in total reserve and light removal (25-33%) partial cuts than in heavy removal (50-85%) partial cuts. Birds were essentially absent from clearcuts in the winter. Almost all of the birds that remain in this area in the winter depend on the forest for feeding and roosting requirements.

MITIGATION OF TIMBER-HARVESTING EFFECTS ON RIPARIAN WILDLIFE

Clearcutting changes the species composition of both bird and small-mammal communities, eliminates most winter bird use, reduces the number of wildlife trees for cavity nesters in the short and long term, and reduces levels of CWD in the short and long term. The effects of riparian forests on the wildlife will be particularly high because the riparian forests of the MSdm1 are higher in bird and small mammal diversity and abundance than upland forests, and they support species that are limited to riparian areas. MSdm1 riparian forests are particularly important to wildlife because they are more similar in tree species composition and structure to old-growth and ESSF forests than to upland lodgepole pine forests.

Conservation of riparian forest bird and mammal species is best achieved by reserving forested habitat in preference to clearcutting. Birds and mammals that are most common in riparian clearcuts (e.g., weasels, Meadow Voles, and Willow Flycatchers) can also be common along small or ephemeral creeks within clearcuts, so that their populations will not be put at risk by reserving forest corridors along the larger creeks, or by any other prevailing forest management schemes. Forest corridor width should be >72 m (36 m on each side of the stream, or 72 m along wetlands or lakes) to be of the greatest benefit to all common birds that require forest habitat. Light partial harvest (16-33% removal) is acceptable in these corridors for most species of birds and mammals, but the effects of partial cutting on cavity nesters will only be minimal if the light cut is "patchy," so that snags do not have to be felled for safety reasons throughout the Riparian Management Zone. Higher levels of partial harvest (>48% removal) do not retain habitat suitability for most forest bird species.

Two species that are fairly common in other biogeoclimatic zones in B.C. (i.e., the ESSF, ICH, and CWH) were rare, and relied on unharvested riparian forests (or corridors >72 m width) in the MSdm1. Reducing reliance of Townsend's Warbler on riparian forests in the MSdm1 over the long term may be as simple as managing for a diversity of tree species in upland areas instead of for pure lodgepole pine forests. Winter Wrens appeared to require CWD and fairly dense undergrowth, as well as continuous forest or wide forest corridors. Clearcutting negates short-term habitat suitability, while only timber-harvesting guidelines that address CWD levels and understorey retention will probably allow for long-term habitat suitability.

Eight bird species that were dependent on riparian habitat did not require forests, but since they nested in cavities and trees, and needed perches, they required some standing live and/or dead trees next to wetlands, lakes, and/or streams.

In the design of corridor reserve networks, riparian corridors (which incorporate upland as well as riparian habitats) are preferred to upland corridors because of the higher numbers and greater diversity of animals, vegetation, and trees that riparian corridors will support.

ASSESSMENT OF FPC RIPARIAN TIMBER-HARVESTING GUIDELINES

Recommendations made in the Forest Practices Code (FPC) *Riparian Management Area Guidebook* (Province of British Columbia 1995a) appear to be much more uniform in application than the high variability of naturally occurring conditions in the Okanagan Highlands. Strict application of the guidelines will result in an upward-branching pattern of linear forest patches along the lower reaches of streams that will lack any connections near the headwaters. In fact, any such guidelines must appear contrived when compared to the variability found in nature. In particular, patches of trees at headwaters would often have escaped fires in deep gullies and on wide, subhydric floodplains that will receive no special consideration in the FPC guidelines, and fires would often have burned across lower stream reaches where there will be reserves of trees.

The situation in headwater streams might be remedied by adding the forested subhydric site series to the definition of stream riparian zone width. (Note that the FPC Wetland Boundary definition already includes the subhydric site series.) It might also be remedied by increasing forest buffer zone widths for floodplain sites where forests are generally shallow rooted and prone to windthrow. The FPC guidelines presently recommend just the opposite on S4 streams, where windthrow-prone sites will be harvested rather than given the 10-m reserves recommended for low windthrow hazard areas under Best Management Practices. Variability closer to natural conditions along lower stream reaches might be added by retaining only enough trees to achieve the overall survival levels naturally found in typical drainages.

These changes would more closely mimic natural variability of live-tree survival after fires, to which it is generally assumed wildlife (including species or groups not examined here) has adapted. However, naturally occurring conditions will not necessarily protect stream function and quality, and may often result in mass sedimentation or debris flows, so that it may be desirable to be "overly protective" of riparian areas to protect watersheds. Along lower stream reaches, the guidelines are presently overly protective of live trees compared to naturally occurring conditions. However, the FPC guidelines should not be changed because timber harvesting will only mimic fires in terms of live-tree retention, but will not mimic natural wildlife tree and CWD cycles upon which many species depend.

The FPC guidelines will benefit wildlife on S1, S2, and S3 streams where cycles of wildlife trees and CWD may more closely mimic natural cycles with tree reserves along streams. Wildlife tree and CWD levels on S4 streams that will not necessarily have a reserve zone and on S5 streams will likely be intermediate between natural levels and the extremely low levels caused by clearcutting. Wildlife tree and CWD cycles on S6 headwater streams (5% maximum live-tree retention) will be well below natural levels.

For most forest-reliant species, habitat suitability will be maintained only along lower stream reaches where reserve zones >40 m wide will be established. For species that only need nesting or perching trees, or stream invertebrates, the FPC guidelines should be adequate in most cases, especially if special attention is given to identifying nest sites of herons and Osprey, which do not necessarily nest immediately beside riparian areas.

FURTHER STUDY

The function of riparian habitats as landscape connections for wildlife may be particularly important. This study documented the downward extension of wildlife habitat from the ESSF through the MSdm1 along riparian forests. Since spruce is a common element of riparian habitats right to the Okanagan Valley bottom, this habitat connection might continue through other biogeoclimatic zones to the valley. Studies on this possible connection could assist in landscape planning for wildlife across large elevational gradients.

Riparian forest only occupies a small fraction of the landscape in the Okanagan Highlands. If riparian forest suitability is compromised for certain species for large portions of the projected 80- to 100-year rotation age, then populations may be too low to permit immigration into habitats that may have regenerated suitably. Landscape-level assessment of the extent of riparian forest, and model development of predictions of suitability for selected species could indicate whether projected rotations may leave some species at risk.

Projections from this study for managed upland forests are for extremely low wildlife tree and CWD cycles that have few natural precedents. These will have much higher potential to be detrimental to a wide variety of species requiring wildlife trees and CWD, simply because the uplands cover much larger areas than riparian forests. The full impact of upland management needs to be factored into any scheme for riparian forest management, since the greatest number of species use both upland and riparian areas. Estimates of the level of these impacts might be predicted by studying wildlife use of the few areas where wildlife trees and CWD are at naturally very low levels.

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1 INTRODUCTION

Riparian areas often support the highest diversity and abundance of wildlife in temperate zones because of their diversity of natural features, because they can support species uniquely adapted to the riparian habitats, and because they can support upland species that may depend on riparian habitats for part of their life cycle. Riparian areas are vital parts of an ecosystem acting as connecting areas for surrounding uplands for water and nutrient flows. Clean, clear water is also essential for healthy fish populations. For these reasons and more, riparian areas are often singled out as needing special protection and management. Rautio and Bunnell (1992) identified the greatest single problem in managing riparian ecosystems in B.C. as the lack of basic information on size, shape, species composition, and ecosystem responses. They stressed the need for baseline research on wildlife habitat, biodiversity and ecosystem functions, and on the impacts of forest management regimes.

The purpose of this overall research project was to identify species for which riparian ecosystems were relatively important, and to provide recommendations for forest management that will best protect these species.

There has been very little research in B.C. on the influence of timber harvesting on the diverse wildlife of riparian areas (Rautio and Bunnell 1992). The objectives of the research phase of this project were first, to describe the difference in riparian wildlife and vegetation of the Okanagan Highlands compared to uplands; second, to describe the effects of clearcutting on riparian and upland wildlife and vegetation; and third, to describe the effects of retention of forest corridors of various widths and levels of partial harvest on riparian wildlife and vegetation. The research examined each of the three objectives for breeding birds and small mammals, and a more limited set of objectives for the natural cycles of trees and coarse woody debris (CWD), for amphibian breeding in forests and clearcuts, and for winter-active wildlife use of riparian zones under different timber harvest treatments.

The objective of the second phase was to develop timber-harvesting guidelines that would maintain the natural diversity of riparian areas in the Okanagan Highlands, and to compare these guidelines to the riparian timber-harvesting guidelines of the Forest Practices Code of 1995.

2 TREES AND COARSE WOODY DEBRIS

2.1 Introduction

Riparian ecosystems have often been singled out as areas needing special protection and management during resource extraction and utilization (e.g., Knopf et al. 1988a) because they often support the highest diversity and abundance of wildlife in temperate areas. Riparian ecosystems can support species uniquely adapted to them as well as provide habitat for wider-ranging species for part of their life cycles. Rautio and Bunnell (1992) identified the greatest single problem in managing riparian ecosystems in B.C. as the lack of basic information on size, shape, species composition, and ecosystem responses. They stressed the need for baseline research on wildlife habitat, biodiversity, and ecosystem functions, and on the impacts of forest management regimes.

The purpose of this overall research project was to identify species for which riparian ecosystems were relatively important, and to provide recommendations for forest management that will best protect these species. This report describes the results of the research in five parts, shows the guidelines for timber harvesting that would maintain the natural diversity of riparian areas in the Okanagan Highlands of the southern interior of British Columbia; and compares these guidelines to those in the Forest Practices Code of 1995.

Spruce and subalpine fir will tend to dominate on wet (hygric and subhydric) sites in the Montane Spruce and Engelmann Spruce–Subalpine Fir biogeoclimatic zones of the southern interior of British Columbia, while lodgepole pine will tend to dominate seral forest stands on drier sites (Lloyd et al. 1990). Fire was the dominant force dictating forest stand age and composition prior to intensive fire-control measures introduced in the second half of the twentieth century. Young stands were dominated by lodgepole pine, which is well adapted to germinating in the post-fire environment, and are slowly converted to spruce and subalpine fir stands in the absence of major fires. This pattern is not unique to the Okanagan Highlands, but is found throughout the lodgepole pine forests of the western cordillera (e.g., Romme and Knight 1981). Although this riparian-upland pattern is obvious in many situations, some doubt remained as to how pervasive or consistent this pattern is, and to the causes of the pattern. Romme and Knight (1981) suggest that riparian forests are most likely to be spruce-dominated because fires tend to

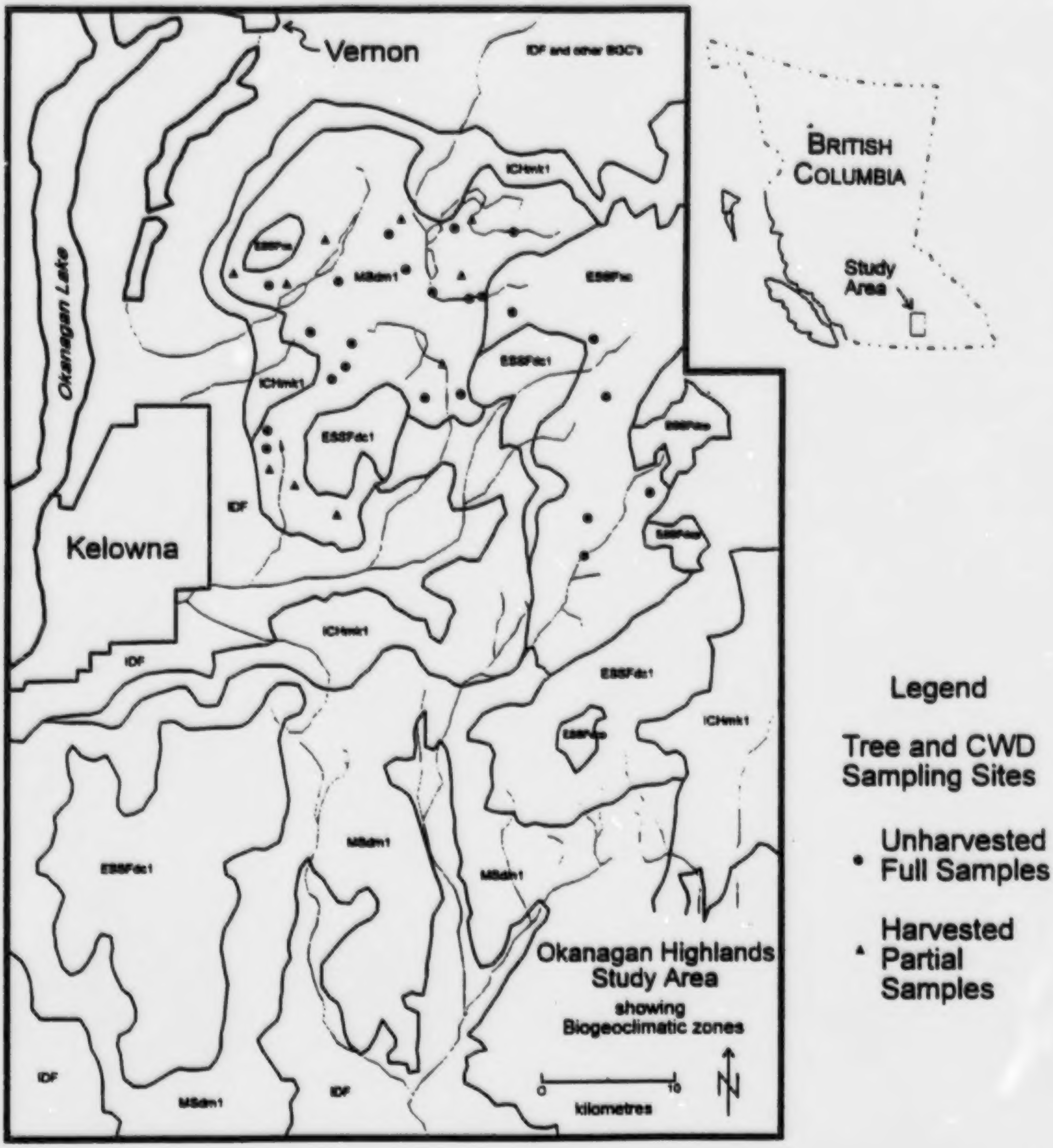

Vernon
IDF and other BGC's
BRITISH COLUMBIA
Study Area
ICHmk1
MSdm1
ESSFxc
ESSFdc1
ICHmk1
ESSFdc1
Okanagan Lake
IDF
Kelowna
ICHmk1
IDF
ESSFdc1
ICHmk1
ESSFdc1
MSdm1
MSdm1
Okanagan Highlands Study Area
showing Biogeoclimatic zones
IDF
MSdm1
IDF
0
10
kilometres
N
Legend
Tree and CWD Sampling Sites
• Unharvested Full Samples
▲ Harvested Partial Samples

Figure 1. Trees and CWD study sites in the Okanagan Highlands of southern British Columbia.

miss the wetter sites or deeper gullies (so that the riparian forests are older) and because "succession following fire is more rapid and/or more direct" on riparian sites.

The purpose of this study on riparian forest structure in the Okanagan Highlands was to determine the relative tree species composition of riparian and upland sites in relation to stand age, and to determine the natural successional paths in the absence of timber harvesting. We also examined two features of stand structure (wildlife trees and coarse woody debris) that are important to wildlife and are generated from the trees of the forest stand. Wildlife trees were defined as standing dead trees. Coarse woody debris was defined as downed logs >7.5 cm diameter.

The specific objectives of this study were to:

- Determine differences in stand age distribution between upland and riparian areas.
- Determine the differences in forest stand species composition and abundance, and the effects of stand age on stand composition in unharvested upland and riparian areas.
- Determine the numbers and species of trees that survived the previous major disturbance events in riparian and upland areas.
- Determine the levels of coarse woody debris and the densities of wildlife trees, as well as their relative use by animals, in relation to stand age and riparian or upland position.
- Develop predictive models for stand structure that will assist in developing recommendations for timber-harvesting guidelines that would retain the diversity of tree-related elements in riparian areas.

2.2 Study Area

The study area was in the Okanagan Highlands east of Kelowna, B.C. (Figure 1). The northern part of the study area was in the Vernon Forest District and the southern part was in the Penticton Forest District, both in the Okanagan Timber Supply Area of the Kamloops Forest Region of the B.C. Ministry of Forests.

This study focused on the two predominant biogeoclimatic zones of the Okanagan Highlands. The Okanagan Dry Mild variant of the Montane Spruce (MSdm1) Zone predominated at medium elevations (1300-1600 m), and the Very Dry Cold subzone of the Engelmann Spruce–Subalpine Fir (ESSFxc) Zone predominated at higher elevations (1600-2000 m) (Lloyd et al. 1990). The forest vegetation of the MSdm1 was dominated by a fire-caused disclimax of lodgepole pine in upland stands, which generally proceeds to a spruce (white-Engelmann hybrid) and subalpine fir climax in the absence of stand-replacing fires. Spruce and subalpine fir were common at all stages of regeneration along creeks. The upland and riparian forests of the ESSF were dominated by Engelmann spruce and subalpine fir, although lodgepole pine was abundant after stand-replacing fires.

2.3 Methods

2.3.1 Study sites

Sampling sites were randomly chosen within the MSdm1 and ESSFxc. Randomly generated points were plotted on 1:50 000 maps of the study area. The nearest point to each of these randomly generated points on a water body or stream was located. Only points that fell on forested streams were selected for sampling. Streams that were surrounded by wetlands (as mapped on 1:15 000 forest cover maps) and lakes were not sampled. Stream reaches shorter than 250 m between wetlands or lakes were not sampled because of the 250-m length of our sample plots.

A target of 18 unharvested sites was set for the MSdm1 and 6 unharvested sites for the ESSFxc. For the MSdm1, this allowed us to determine if our sample of unharvested streamside and upland forests was similar to a completely random sample, or whether previous selection of sites for timber harvesting may have introduced some bias. Sites where clearcutting or partial timber harvesting had taken place were not fully sampled. Only the stand age prior to timber harvesting was determined and then extrapolated to the present. To achieve 18 fully sampled sites in the MSdm1, 29 sites were visited. Only 17 MSdm1 sites were used in the final analyses because one site was found to be floristically more like the ICHmk1 than the MSdm1. It was in a deep valley (Darley Creek) where the BGC maps placed the ICHmk1/MSdm1 boundary down low across the valley, but where it should have contoured up the valley to reflect local conditions more accurately.

2.3.2 Sampling methodology

One-hectare sampling plots were centred on the randomly selected stream point (Figure 2). Plots extended 125 m up and down the stream, and 20 m to each side of the stream bank. Four 0.04-ha (40 × 10 m) subplots were placed on the side of the stream nearest to the randomly generated point from which the streamside point had been selected. One edge of each subplot was the stream bank, with the plot extending 10 m upslope. The four subplots were centred on the randomly selected point with a 10-m spacing between subplots.

Each streamside plot was paired with a parallel upland plot (Figure 2). From the randomly selected streamside point, 100 m was measured perpendicular to the stream direction in the direction of the randomly generated point. The 100-m distance was only extended when the stream reach was curved so that the plot had to be further away to keep the ends of the upland and streamside plots from overlapping, or when the valley bottom was wider than 100 m. Only in one case was the distance less than 100 m (75 m), this was to prevent the upland plot from extending into a clearcut. Sampling within the upland plot was the same as in the streamside plot, except that an arbitrary centreline parallel to the stream direction was established instead of using the stream as a centreline.

The physical characteristics of each site were measured. Stream order was assessed and stream gradient measured from 1:50 000 topographic maps. On site, stream width and "active floodplain" width were measured at six points separated by approximately the bank-to-bank width. "Active floodplain" is defined in the *Riparian Management Area Guidebook* (Province of British Columbia 1995a) of the Forest Practices Code of British Columbia (hereafter referred to as FPC riparian guidelines) as the approximate equivalent of annual floodplain or active channel. We also measured the width of what might be more usually thought of as the floodplain: the hygric or subhydric site series (Lloyd et al. 1990) adjacent to a stream where trees grow primarily on organic hummocks with water standing in the pools between the hummocks. The height and distance profile from the randomly selected point on the stream to the upland sample point was measured to assess gully depth.

In each unharvested plot, the time since the last major disturbance event was determined by coring trees from the dominant cohort of canopy trees, and was termed "stand age." At every site <250 years old, fire could be identified as the previous major disturbance event that had given rise to the dominant cohort of canopy trees. Possible veteran trees that may have survived the last major disturbance were identified and also cored to determine if they were veterans. At most sites, a diameter limit could be found that differentiated veterans from trees that had grown since the last major disturbance. In some cases, small patches of veteran trees were found. In these cases, numerous trees were cored to determine the boundaries of the patch. Within the patch, only trees older than the previous major disturbance were termed veterans. The 1-ha plots were then fully censused for veteran trees. All living trees >10 cm diameter breast height (dbh) were tallied in the four 0.04-ha subplots. Data collected for each tree were: species, canopy position or strata (veteran tree, over-canopy tree, canopy tree, or sub-canopy tree), and dbh to 10-cm classes (10 to 19.9 cm, 20 to 29.9 cm, etc.).

Wildlife trees (dead standing trees) >30 cm dbh were tallied in the 1-ha plots. All wildlife trees >10 cm dbh were tallied in the four subplots. Only wildlife trees of classes 3 to 7 (Province of British Columbia 1995b) were counted. Dead trees <0.5 m height were not counted, nor were dead trees <1.0 m height that had no part of the outer wood surface (the surface immediately under the bark) remaining intact. These were considered to be class 8 wildlife trees and were not tallied. Data collected for each tree were: species, dbh to 10-cm classes, height, wildlife tree class, probable cause of tree death if readily identifiable, and signs of animal use. Animal species using the tree were noted where possible, and time since use was estimated as recent or older. Recent use was characterized by freshly exposed wood, and probably indicated use within the previous 8-12 months.

Downed logs, or coarse woody debris (CWD), ≥7.5 cm diameter were tallied on a 90-m intercept line centred at the randomly selected point. The line was parallel to the stream bank and 5 m distant. Diameter, height, and decay class (1 to 4 of Maser et al. 1979) of

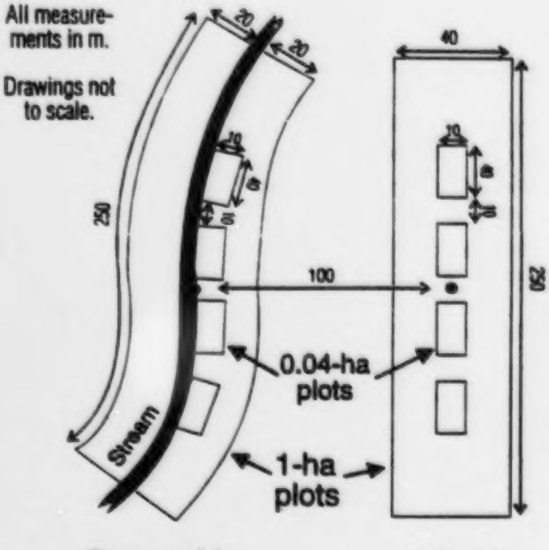


Figure 2. Typical layout of paired streamside and upland sampling plots and subplots.

each piece intercepting the line were noted. The source of each piece of CWD, i.e., developed from trees grown in the pre-fire stand or from trees grown after the last major fire, was estimated.

2.3.3 Data analyses

The stand age distribution of the unharvested sampled sites within the MSdm1 were compared to the stand age distribution of the clearcut and partially cut sites to determine if there were any uncontrolled selection biases within the unharvested site data set. The stand age distributions of the streamside and upland stands were compared.

Live tree density estimates (trees/ha) were calculated for each site for each species, strata, and dbh class. Very few over-canopy trees were counted, and determining these trees in streamside and old-growth stands was often problematic because of the uneven height of the canopy. For analysis, over-canopy trees were grouped with the canopy trees. Densities of veteran trees were based on the 1-ha plots, whereas all other densities were based on the four 0.04-ha subplots at each site. One-factor Analysis of Variance (ANOVA) within the MSdm1 and within the ESSFxc was used to compare tree densities across the topographic position (streamside/upland) factor broken down by stand age, species, and strata. Mean tree size (dbh) of the canopy layer was also calculated for each site, and for each species at each site, and compared between upland and streamside sites. However, because size was expected to vary with stand age, Analysis of Covariance (ANCOVA) methods were used for this comparison. To examine total tree structure of the stand, the ratio of canopy trees (including veterans) to sub-canopy trees was calculated and examined for topographic position and BGC differences using ANOVA. Probability of a Type I error (α) for all analyses was set at 0.05. Where variances of groups within a comparison were significantly different, equivalent non-parametric procedures were applied. The results of these non-parametric tests were only presented when the conclusion of the test, i.e., acceptance or rejection of the hypothesis, was different from the parametric test. When presented within the text, all means are followed by the standard error.

Veteran trees were impossible to quickly distinguish in stands >250 years old because of potential size overlap with younger trees. Therefore, stands >250 years old were left out of analyses involving veteran trees. Veteran trees could also not be reliably identified in the ESSFxc. The ESSFxc stands were simply not as even aged as the MSdm1 stands, so that our fairly crude methods to determine both stand age and what constituted veteran trees were inadequate. Coring each tree within these stands to construct accurate stand-age tables was not possible within the allotted time or budget. In the MSdm1, however, any veteran tree that was questionable was cored to determine its age accurately.

The density of veteran trees in an existing stand will always be less than the number that actually survived the previous disturbance if any of them die or fall between the time of disturbance and the sampling. In this analysis in the MSdm1, stand age was 54-189 years after disturbance so that many of them may have indeed fallen. Based on stage of decay, extent of burn scarring, and time since fire, it was sometimes possible to determine if wildlife trees and CWD had survived the disturbance event and only died later. For all stands <250 years old, these additional trees were added to the tally of living veteran trees to estimate the density of veteran trees immediately after the time of disturbance. Veteran tree distribution within plots was estimated, and the most likely causes for the survival of veteran trees were examined for each site.

Standing dead tree (wildlife tree) density estimates (trees/ha) were calculated for each site for each species and for the 10 to 29.9 cm and >30 cm dbh classes. The density estimate for small wildlife trees (10 to 29.9 cm) was based on the four 0.04-ha subplots, while for large wildlife trees (>30 cm) they were based on the 1-ha plots. Density of wildlife trees showing new woodpecker or cavity-nester use was calculated for the small and large trees separately, and percentage of each class showing use was also calculated. Because wildlife tree densities could be expected to vary with stand age, pairwise *t*-tests were used to compare upland and streamside sites to minimize any possible bias associated with stand age. Use was also calculated for each species and each size class of trees, and the rate of tree use was compared using χ^2 contingency tables.

To help develop a predictive model of wildlife tree density, wildlife trees were categorized as developing from within the present stand, or being remnants of the previous stand. Wildlife trees with extensive burn damage, or that were larger than any living trees in the post-disturbance stand, were considered to be from the previous stand. The development of wildlife trees from within the post-fire stand was analyzed using ANCOVA, with stand age as the covariate. The number of wildlife trees remaining from pre-fire stands was also analyzed using ANCOVA, with topographic position and previous stand age as grouping factors and stand age as a covariate.

Volumes of downed CWD/ha in m^3/ha were estimated using the formula from Harmon et al. (1986): Volume = $\pi^2 \sum d^2 / 8L$, where d=log diameter and L=transect length. Because CWD could be expected to vary with stand age, pairwise *t*-tests were used to compare upland and streamside sites to minimize any possible bias associated with stand age. To help develop a predictive model, CWD volumes developed from trees grown in the pre-fire stand were estimated separately from CWD developed from trees grown after the last fire. CWD volume developed from within the post-fire stand was analyzed for topographic position differences using ANCOVA, with stand age as the covariate.

2.4 Results

2.4.1 Stream physical characteristics

In the MSdm1, sampled streamside sites were all on first- or second-order streams. Stream gradients ranged from 2-7% and did not vary significantly with stream order ($F_{1,15}$=0.04, P=0.84). Bank-full widths of first-order streams were significantly narrower than second-order streams (1.27±0.19 m, N=9 vs. 3.68±0.37 m, N=8; $F_{1,15}$=35.95, P=0.000). The range of first-order stream widths was from 0.53 to 2.52 m, while the range of second-order streams was from 1.95 to 4.76 m. Mean stream gradient in the ESSFxc (8.7±1.2%, N=6) was significantly higher ($F_{1,21}$=29.60, P=0.000) than in the MSdm1 (3.8±0.4%, N=17). While forested stream reaches in the ESSFxc were significantly steeper than forested streams in the MSdm1, this would not likely apply to a random sample of all streams. Many headwater streams in the ESSFxc had very low stream gradient, but were usually surrounded by marshes, fens, and shrub swamps rather than forests, and were consequently not sampled here. Two first-order streams, two second-order streams, and two third-order streams were sampled in the ESSFxc. First- and second-order streams were of similar sizes to the MSdm1, but third-order streams were much wider (7.8 to 11.0 m, bank-full width).

Only three valleys in the 23 stream sites sampled had gullies deeper than 20 m (range 20 to 32 m). The rest of the valleys were shallow enough that they were not considered gullies. The mean of the remaining valley depths was 5.9±0.8 m (N=20).

Bank-full stream widths were, on average, 84% of the "active floodplain" widths as defined in the FPC riparian guidelines. Under that definition, an active floodplain is the approximate equivalent of annual floodplain or active channel, and is therefore not very different from bank-full width. However, stream width (r^2=0.00), "active floodplain" width (r^2=0.00), stream gradient (r^2=0.06), and gully depth (r^2=0.02) were all very poor predictors of the actual extent of riparian forest, i.e., the floodplain width as defined by the presence of standing water and trees growing on organic hummocks. In the MSdm1, the sizes of gullies and valleys often bear little relationship to the size of the stream within them because many of the valleys are remnants of glacial meltwater channels. The only sure way to determine the extent of riparian forest is to visit the site in the snow-free season (although aerial photo interpretation can also be useful).

In the MSdm1, site series 07 (Spruce–Trapper's tea–Horsetail) of Lloyd et al. (1990) provided a very good description of the floodplain forests found. Presence of common horsetail (*Equisetum arvense*) in the understorey at coverage levels of >5% (as in the key of Lloyd et al. 1990) was found to be the most reliable indicator of this hygric to subhydric site series. The non-forested hydric site series 08 (Willow–Sedge) was occasionally found in small pockets within the sampled floodplains, but for the most part, areas where site series 08 predominated were not considered forests, and were therefore not within the selection and sampling regime of this study.

2.4.2 Stand age distribution

In the MSdm1, the age distribution of the 17 unharvested sites and the 11 partially cut or clearcut sites was very similar (Table 1). The mean stand ages were not significantly different nor were the medians, which were significantly skewed to the left of the mean of each distribution. The largest number of samples were in the 80 to 120-year age class, which would translate to major fires in the period between 1875 and 1915. The left tail of the unharvested distribution was bounded by a stand age of 54 years, because there have been no fires of any significant size in the last 54 years in the study area so none were available for sampling. No partially cut or clearcut sites with a stand age of <85 years were found, because forests <85 years are generally not considered for timber harvest. The stands <85 years old accounted for only 11% of the total sample but 18% of the unharvested sites. The analyses (when unadjusted for stand age) proceeded in the knowledge that the unharvested sites may not be completely representative of the full range of forests that would have been present in the absence of timber harvesting, and that the potential bias might be up to 7% toward younger age classes.

Table 1. Stand age distribution of unharvested and harvested (partially cut or clearcut) randomly selected upland forest stands in the MSdm1.

Stand age (years)	Number of sites Unharvested	 Harvested[1]
50-59	2	0
60-69	0	0
70-79	1	0
80-89	3	2
90-99	1	1
100-109	1	2
110-119	3	0
120-129	0	1
130-139	0	1
140-149	1	0
150-159	0	0
160-169	1	1
170-179	1	0
180-189	1	0
190-199	0	1
200-249	0	1
>249	2	1
Total	17	11

Statistic	Unharvested	Harvested	Distributions compared
Mean	140.6	147.5	$F_{1,26}$=0.05; P=0.83
Variance	8453.9	4668.0	$F_{16,10}$=1.81; P=0.31
Median	111	122	$T_{17,11}$=76.5; P=0.42
Skewness	1.86 P=0.003	1.32; P=0.05	
Kurtosis	3.00 P=0.06	1.13; P=0.32	
Kolmogorov-Smirnov normality test	0.244 P<0.05	0.198 P>0.10	

[1] Age of clearcut "harvested" sites adjusted to 1995, based on age of tree at harvest + years since clearcutting.

The streamside and upland stands were the same age at all but 3 of the 23 stands sampled. In the MSdm1, no significant differences could be shown between the streamside and upland stand age distributions when only 18% (3 of 17) of the paired sites differed (e.g., for mean stand age, $F_{1,32}$=0.10, P=0.75). At those three stands, the streamside stands had escaped the last fire that established the dominant cohort of trees in the upland stand. In each of the three cases, veteran trees that had escaped the last fire in the upland were of the same age as the streamside stands. These three sites were associated with the deepest gully sampled (28 m) and the widest floodplains sampled (55 and 70 m). Therefore, within the MSdm1, 18% (3 of 17) of the "spruce in riparian vs. lodgepole pine in the upland" pattern might be explained simply by the existence of topographic features that affect fire behaviour. These are simply older stands where the natural successional patterns toward spruce and subalpine fir are further advanced. However, riparian spruce forests occurred along more than 18% of the total stream reaches in the study area, so that topography alone was not the only explanation of the riparian spruce vs. upland pine pattern.

2.4.3 Live trees

There were only three common species of trees (lodgepole pine, spruce, and subalpine fir) making up 97% of the 8754 trees counted in the plots. Both Engelmann spruce and Engelmann-white hybrid spruce occur in the area, and no attempts were made to separate spruce trees to species. Other species encountered uncommonly were (in order of decreasing abundance): Douglas-fir, alder, western redcedar, Scouler's willow, trembling aspen, and black cottonwood. Western larch trees do occur very uncommonly in the study area, but none were in the sample plots. For the analyses, these uncommon species have been grouped.

The stands were aged using lodgepole pine as the "cohort" tree whenever possible, since they establish very quickly, and are fully established within the first decade after major fires (Johnson and Fryer 1989). Stand ages derived from spruce in this study were simply not as reliable, and were usually younger than the ages derived from lodgepole pine. Spruce establishes predominantly within the first decade after fire, but significant recruitment (i.e., 1000 individuals/ha) can continue for the first three decades after fire, with sporadic recruitment thereafter (Johnson and Fryer 1989).

Sites were grouped according to BGC and stand age for analysis of live-tree density variation with topographic position. Within young stands <192 years old, no significant correlation of densities of any canopy tree species could be shown with stand age. Therefore, these stands were grouped for analysis and termed young stands. Old-growth stands were >275 years old. There were no sampled stands between 192 and 275 years old.

In young MSdm1 stands (Table 2), the density of lodgepole pine in the canopy layer was significantly higher in the upland sites than the streamside sites, and the density of spruce and subalpine fir was the inverse, being significantly higher in streamside sites. There were no significant differences in the sub-canopy layer, where spruce was the most common species, followed by subalpine fir. In young ESSFxc stands, subalpine fir was more abundant in the canopy layer in upland stands than in streamside stands. Upland stands were a mix of spruce, subalpine fir, and lodgepole pine (in order of decreasing abundance), while streamside stands were dominated by spruce in the canopy layer. There were no significant species differences in the sub-canopy layer, with approximately equal densities of subalpine fir and spruce in both upland and streamside sites. In old-growth stands, there were no significant topographic position differences in any species densities within either the canopy or sub-canopy layers. These sites were dominated by subalpine fir and spruce.

Tree species densities were compared among the four basic groups of sites identified in the topographic position analysis of the preceding paragraph, i.e., young MSdm1 streamside, young MSdm1 upland, young ESSFxc, and all old-growth stands (Table 3). Canopy-layer lodgepole pine was most abundant in young MSdm1 stands, less abundant in young MSdm1 streamside stands and young ESSFxc stands, and very uncommon in old-growth stands. Lodgepole pine predominated (>70% of trees by stem count) in all but two of the upland MSmd1 stands younger than 192 years. In those two stands, lodgepole pine only made up 30-40% of the stands. One of the stands was dominated by Douglas-fir, and the other was 160 years old and was spruce-dominated, probably because it was on a northerly aspect and slightly cooler and wetter than other sites. All upland stands in both the MSdm1 and ESSFxc younger than 192 years contained lodgepole pine with one exception: a 104-year-old stand in the ESSFxc that, based on log sizes and wildlife tree sizes, was old growth when it burned. There was probably no lodgepole pine in the stand when it burned so that no seed source was available and lodgepole pine could not establish in the post-fire stand. In the single upland stand sampled that was 275 years old, as much lodgepole pine was found standing dead as alive (31/ha). In older stands, lodgepole pine was extremely uncommon or entirely absent.

Canopy-layer spruce was most abundant in young MSdm1 streamside stands and on young ESSFxc sites, with lower densities in old-growth stands, and lowest densities in young MSdm1 upland stands (Table 3). Canopy-layer subalpine fir was equally abundant in all groups except young MSdm1 upland stands where it was very uncommon (just 17/ha). In the sub-canopy layer, spruce was equally abundant in each group (Table 3), although subalpine fir was significantly more abundant in old-growth and young ESSFxc stands than in young MSdm1 stands. No lodgepole pines occurred in the sub-canopy layer because they only establish after major fires.

The ratio of canopy to sub-canopy trees for young MSdm1 stands was significantly higher in upland sites than streamside sites (Table 2) because many of the upland sites were even-aged pine stands with virtually no understorey trees, and because streamside sites had significantly lower total canopy densities ($F_{1,29}$=4.19, P=0.05). In young ESSFxc stands and in all old-growth stands, the ratio of canopy to sub-canopy densities was not significantly different in streamside and upland sites, and the ratios were much closer to 1:1. In the comparison of the four groups of sites, young MSdm1 upland stands had the highest canopy to sub-canopy ratio, young MSdm1 streamside sites and young ESSFxc sites were lower, and old-growth stands had the lowest ratio (Table 3). The general trends of succession indicated by both the canopy layer densities and canopy to sub-canopy ratios were for young lodgepole pine or lodgepole pine-spruce stands to become subalpine fir-spruce stands over time in both the MSdm1 and ESSFxc.

Within the cohort of canopy trees grown after a fire in the MSdm1, mean tree size was significantly correlated with $\log_{10}$(stand age) (r^2=0.603, $F_{1,31}$=47.10, P=0.001). The slopes of the streamside and upland regressions were not significantly different ($F_{1,32}$=0.01, P=0.92), so that ANCOVA could be used to compare mean tree size correcting for stand age. Mean tree size in the streamside stands was significantly larger ($F_{1,32}$=14.94, P=0.001) than in upland stands, with the streamside stands averaging 5.0 cm larger in mean dbh at stand ages between 100 and 120 years. This size difference held even into the old-growth stands. The mean size of canopy trees in the streamside MSdm1 sites was described by the regression: Mean dbh = 24.18 × $\log_{10}$(stand age) – 24.50; r^2=0.66. The mean size of canopy trees in the upland MSdm1 sites was described by the regression: Mean dbh = 24.81 × $\log_{10}$(stand age) – 30.76; r^2=0.72.

Breaking down the MSdm1 canopy cohort into species, mean dbh of canopy spruce was 5.0 cm larger in streamside stands than in upland stands at ages between 100 and 120 years (streamside mean dbh = 24.6 × $\log_{10}$(stand age) – 22.9, r^2=0.66; upland mean

Table 2. Streamside to upland comparisons of live tree densities by BGC, stand age, canopy position, and species, and the ratio of canopy to sub-canopy trees by BGC and stand age.

Strata Species	Density of live trees Streamside Mean	 S.E.	 Upland Mean	 S.E.	Statistics $F^{1,2}$	 P
Young, MSdm1, <192 years, N=15 pairs						
Canopy						
Spruce	360	62	77	32	16.31	0.000
Subalpine fir	92	32	16.9	5.7	5.22	0.030
Lodgepole pine	291	78	1159	197	16.77	0.000
Other species	6.6	3.5	8.5	7.7	0.05	0.82
Sub-canopy						
Spruce	206	57	114	30	2.06	0.16
Subalpine fir	104	28	63	21	1.34	0.26
Other species	43	22	21	16	0.69	0.41
Canopy to sub-canopy tree ratio	3.4	0.6	28.3	14.0	U=51	0.01
Young, ESSFxc, <130 years, N=4 pairs						
Canopy						
Spruce	323	67	314	124	0.00	0.95
Subalpine fir	83	14	264	88	U=1	0.04
Lodgepole pine	44	31	102	43	1.18	0.32
Other species	0	-	0	-	-	-
Sub-canopy						
Spruce	230	66	191	84	0.13	0.73
Subalpine fir	173	69	209	46	0.19	0.68
Other species	1.6	1.6	1.6	1.6	0.00	1.00
Canopy to sub-canopy tree ratio	1.3	0.2	1.8	0.2	2.37	0.18
Old-growth, MSdm1 and ESSFxc, >275 years, N=4 pairs						
Canopy						
Spruce	181	25	156	46	0.23	0.65
Subalpine fir	203	75	192	57	0.01	0.91
Lodgepole pine	1.6	1.6	12.5	7.7	1.96	0.21
Other species	0	-	1.6	1.6	-	-
Sub-canopy						
Spruce	83	30	72	31	0.06	0.81
Subalpine fir	231	37	275	14	1.21	0.31
Other species	1.6	1.6	3.1	3.1	0.20	0.67
Canopy to sub-canopy tree ratio	1.2	0.3	1.1	0.3	0.13	0.73

[1] 1 num d.f, 28 denom. d.f. for MSdm1 <192 years old; 1 num. d.f., 6 denom. d.f. for other groups.

[2] Where variances were significantly different, and the non-parametric Mann-Whitney Test result was different from the parametric ANOVA, the Mann-Whitney U Statistic and its probability are given.

Table 3. Comparison of live tree densities between four groups of sites in the MSdm1 and ESSFxc.

	Mean density of live trees[1]				Statistics		
	<250 years			>250 years			
	MSdm1		ESSFxc	MSdm1 + ESSFxc (N=8)	Homogeneity of variances[2] P	Kruskal-Wallis	
Strata Species	Streamside (N=15)	Upland (N=15)	All (N=8)			T	P
Canopy							
Spruce	360^c	77^a	319^{bc}	169^{ab}	0.000	19.96	0.000
Subalpine fir	92^b	17^a	173^b	198^b	0.001	21.64	0.000
Lodgepole pine	291^b	1159^c	73^{ab}	7.0^a	0.000	27.14	0.000
Sub-canopy							
Spruce	206	114	210	77	0.003	6.69	0.08
Subalpine fir	104^a	63^a	191^b	253^b	0.26	$F_{3,42}$= 8.77	0.000
Canopy to sub-canopy tree ratio	3.4^b	28.3^c	1.6^{ba}	1.1^a	0.000	20.54	0.000

[1] Numbers in the same row with the same superscript letter were not significantly different at P=0.05.

[2] Bartlett-Box test for homogeneity of variances; where not significant, ANOVA was used rather than Kruskal-Wallis.

dbh = 27.4 × $\log_{10}$(stand age) − 33.5; r^2=0.48; ANCOVA $F_{1,28}$=5.50, P=0.026). No significant difference could be shown in streamside and upland lodgepole pine mean tree size (ANCOVA $F_{1,27}$=1.96, P=0.17), although they did significantly increase with stand age (mean dbh = 34.01 × $\log_{10}$(stand age) − 47.96; r^2=0.60, $F_{1,27}$=42.94, P=0.000). Mean size of canopy subalpine fir was not significantly correlated with $\log_{10}$(stand age) (r^2=0.08, $F_{1,23}$=1.96, P=0.16). Subalpine fir appeared to be slower in gaining height than either lodgepole pine or spruce, so that by the time they became canopy trees they were virtually fully grown and put on little more girth. The larger mean tree size in streamside sites can therefore be largely attributed to the faster-growing habits of spruce on these sites.

2.4.4 MSdm1 veteran trees

On average, 87% of the veteran trees that survived the stand-initiating fires appeared to be still alive at the time of our sampling, 54 to 179 years after the fires. No additional veteran trees were tallied using log or dead tree counts in stands where there were no living veteran trees. All of the following analyses of veteran tree densities and distributions included both the living veteran trees and any trees that were dead in 1995, but that could be identified as having survived the fires that initiated the present stand.

Three of the 15 upland stands were much younger than the adjacent streamside stands, so that comparisons of "veteran" trees between upland and streamside sites were adjusted to the older streamside stand age for this portion of the analysis. Veteran trees were found in all three of these upland stands, but in two of them, the only veteran trees were trees that had established at the same time as the streamside stands. The last fires had missed these two streamside sites, and left high densities of spruce and pine (92.5 to 112.5/ha) surviving in the upland stands. In the third site, the veterans were large Douglas-fir on a rocky outcrop 28 m above the creek bottom. These Douglas-firs had survived the previous two fires and were veterans of the same fire that had resulted in stand initiation at the streamside in the gully below.

Using the streamside stand ages as the baseline age, mean veteran tree densities were not significantly different (P=0.14) between upland (4.5±3.2/ha) and streamside (14.1±5.1/ha) sites. However, the distributions were significantly non-normal (i.e., many sites with no veterans), so that the mean did not describe the distributions well. If we use the upland stand ages as the baseline age, then the sites where the streamside stands were older were entirely composed of veteran trees (Table 4).

Complete survival of trees at streamside sites in the MSdm1 were associated with deep gullies (>28 m at one site) and with wide floodplains (>55 m at two sites). Survival of some veterans (14/ha) was associated with the only other deep gully (20 m) sampled. Setting aside the two gullied sites, density of veteran trees was most highly correlated with the cube of floodplain width (density = 3.03 + 0.003 [floodplain width]^3; r^2=0.96, $F_{1,11}$=283.3, P=0.000, N=13). At a floodplain width of 10.5 m, the predicted density of veterans was zero (i.e., the *x*-intercept). Stratified sampling would be required to better determine the actual strength of this correlation because only two sites in this regression had floodplains wider than 20 m, and both of these sites also had complete survival of trees.

Upland veteran trees were found at 6 of the 15 MSdm1 sites. Two sites were in wet, low-lying ground, two sites were near to streams with wide floodplains that appeared to be the boundaries of large fires, one site was on a rocky ridge top where a large Douglas-fir survived a fire, and one site was where a fire was very patchy in both the upland and streamside areas for unknown reasons.

Table 4. Density and dispersion of veteran trees surviving stand-replacing fires along streams and in upland areas in the MSdm1.

Density and dispersion of veteran trees	% streamside length[1]	% upland area
In patches larger than 1 ha	20	N/A[2]
In patches 0.25-0.75 ha	8	4
Many scattered trees (90-115/ha)	Not found	10
Few scattered trees (10-18/ha)	13	Not found
Very few scattered trees (1-5/ha)	17	13
Number of veteran trees	42	73

[1] Streamside length is based on an area 20 m to each side of stream bank.

[2] Not applicable, because the stand age of upland 1-ha plots was used as the baseline to define veteran trees.

2.4.5 Wildlife trees

Most of the wildlife tree use noted was from woodpeckers scaling or flaking bark to obtain insects or drilling shallow holes exploring for insects. Only one nest hole was found in the 1311 small (10 to 29.9 cm dbh) wildlife trees sampled (0.08% used for nesting), while 24 nest holes were found in the 602 large (≥30 cm dbh) trees sampled (4.0% used for nesting). This will be an underestimate of nesting because nest holes far up the tree stem may easily have been missed.

Only the work of Pileated Woodpeckers was completely diagnostic to species. Although sapsucker wells are distinctive, they are only drilled in live trees, so even when they were detected in our dead tree sampling, they were not made after the tree had died. Very few (0.8%) small wildlife trees were recently used by Pileated Woodpeckers compared to 6.1% of large wildlife trees (χ^2=35.14, P=0.000) in the MSdm1. No recent Pileated Woodpecker use was found in the ESSFxc, although one old working was found in a large wildlife tree in one sample.

There were no significant differences between streamside and upland sites in densities of small wildlife trees in either the MSdm1 or ESSFxc (Table 5). Large wildlife trees were significantly more abundant in streamside sites than in upland sites in the MSdm1, but there were no significant differences in the ESSFxc. The density of large wildlife trees recently used by birds was also significantly higher in the streamside sites than in the upland forests of the MSdm1. However, since the percentage of large wildlife trees recently used (only considering stands where they were available) was not significantly different, the birds appeared to be using the large trees as they were available, and there were simply more available in the streamside forests than in the upland forests.

In the MSdm1, spruce and subalpine fir wildlife trees were significantly more abundant at streamside than upland sites, while lodgepole pine wildlife trees were significantly more abundant at upland sites (Table 5). This reflected the live tree stand composition from which the dead trees were recruited — live spruce and subalpine fir were more abundant at streamside sites, and lodgepole pine was more abundant in upland sites. No species of wildlife tree was significantly more abundant in streamside than upland sites in the ESSFxc.

Large wildlife trees of each species received significantly higher rates of recent usage by woodpeckers than small wildlife trees. For each species, recent use of large wildlife trees was about double the use of small wildlife trees. For subalpine fir, use of small trees was 5.6% compared to 10.7% for large trees (χ^2=6.33, P=0.01, N=482 small trees and 263 large trees); for lodgepole pine, 23.3% compared to 62.6% (χ^2=78.9, P=0.000, N=602 small trees and 131 large trees); and for spruce, 33.3% compared to 51.8% (χ^2=11.68, P=0.001, N=207 small trees and 137 large trees). No small Douglas-fir wildlife trees were found in the sampling, but 42% of 43 large

Table 5. Paired streamside to upland comparison of wildlife tree densities in the MSdm1 and ESSFxc.

	MSdm1 (N=17 pairs)				ESSFxc (N=6 pairs)			
	Streamside	Upland			Streamside	Upland		
Parameter	Mean S.E.	Mean S.E.	t	P[1]	Mean S.E.	Mean S.E.	t	P
SMALL (<30 CM DBH) WILDLIFE TREES								
Density	142.3 17.5	185.7 43.0	1.07	0.30	142.7 34.2	287.5 113.4	1.53	0.19
Density recently used	22.4 4.3	45.6 11.4	1.95	0.07	29.2 15.1	26.0 10.4	0.65	0.54
% recently used	15.4 3.2	26.2 6.4	1.85	0.08	18.6 8.1	14.1 4.7	0.59	0.58
LARGE (≥30 CM DBH) WILDLIFE TREES								
Density	14.6 3.3	7.4 3.4	3.34	**0.00**	17.3 6.2	21.2 6.0	0.57	0.60
Density recently used	7.1 1.6	2.8 1.0	2.67	**0.02**	1.8 0.9	3.7 0.8	1.12	0.31
% recently used	49.5 7.1	57.0 10.0	0.71[2]	0.50	6.9 3.5	30.8 11.7	1.65	0.16
DENSITY BY SPECIES (ALL SIZES)								
Spruce	43.4 9.6	12.0 5.2	2.87	**0.01**	45.9 19.8	27.5 13.3	0.77	0.46
Subalpine fir	55.0 13.8	14.8 7.0	2.60	**0.02**	99.3 30.0	248.0 117.5	1.50	0.25
Lodgepole pine	51.7 12.7	165.0 45.4	2.41	**0.03**	7.7 4.9	27.7 12.3	1.51	0.16

[1] Significant P denoted in **bold**.

[2] N=11 in paired test, since only sites where wildlife trees >30 cm dbh occurred in both streamside and upland sites were considered.

Douglas-fir wildlife trees had recently been used by woodpeckers. Among small wildlife trees, spruce was used significantly more frequently than lodgepole pine, and both were used significantly more frequently than subalpine fir. Among large wildlife trees, subalpine fir was used significantly less frequently than spruce, lodgepole pine, and Douglas-fir, among which there were no significant differences in recent use. Overall woodpecker usage (recent and older use) of large wildlife trees was 24.0% for subalpine fir, 68.7% for lodgepole pine, 68.7% for spruce, and 88.1% for Douglas-fir. Overall woodpecker usage of small wildlife trees was 10.2% for subalpine fir, 27.1% for lodgepole pine, and 42.0% for spruce.

After a stand-replacing fire, the density of wildlife trees will be approximately equal to the total tree density that existed prior to the fire. Over time, these trees will begin to fall, so that eventually, none of them will remain standing. No samples were available from stands between 0 and 54 years, so that a survival function of standing dead trees after a fire could not be reliably estimated. Within the stands that were studied, virtually all small trees (10 to 29.9 cm dbh) killed in the stand-initiating fires had already fallen over and were counted only as CWD. In stands >114 years, no small wildlife trees remained from the pre-fire stand. For the 26 stands between 54 and 114 years of age, only seven stands had any small wildlife trees remaining from the pre-fire stand. Overall, the mean number of small wildlife trees remaining from the pre-fire stand was 1.92±0.67/ha (N=26). There was no significant difference between streamside and upland stands ($F_{1,24}$=0.87, P=0.36).

While small wildlife trees tended to fall over from the base within a few decades of a fire, many larger trees (≥30 cm dbh) killed in fires appeared more likely to break off at points further up the bole, and consequently remain standing as wildlife trees for longer periods. The oldest stand was 140 years, in which we found wildlife trees still standing after being killed in the stand-initiating fire. There were 15 pairs of stands in the MSdm1 and ESSFxc between 54 and 140 years old (with one stand omitted from this analysis because it was the only stand where Douglas-fir had been abundant in the previous stand). Time since fire (stand age) was not a significant covariate within this range of stand ages (Table 6), but was obviously important, since no wildlife trees were found that remained standing for 140 or more years after they had been killed by fire. Within this range of stand ages, the density of large wildlife trees remaining from the previous stand was affected by both age of the previous stand and topographic position — more large wildlife trees were found at streamside sites and where the previous stand age had been >150 years (Table 6). Both stand age and streamside position have already been shown here to affect live tree size, and these effects appear to also extend to the density of dead standing trees even long after the stand has been killed by a fire.

Development of small wildlife trees (Figure 3A) from within the post-fire stand was very rapid, with densities reaching maximum levels at stand ages of 75 years, and no significant correlation with stand age (r^2=0.02, P=0.32). Development of large wildlife trees from within the post-fire stand can only begin as trees reach large sizes. A significant threshold appeared at stand ages of about 125 years, when large wildlife trees that developed within the stand started to appear at densities higher than 10/ha. This density continued to significantly increase with stand age (r^2=0.65, P=0.000; Figure 3B). For wildlife trees developed from within the post-fire stand, no significant differences could be shown between streamside and upland stands ($F_{1,44}$=0.36, P=0.55) or between the MSdm1 and ESSFxc ($F_{1,44}$=0.08, P=0.77).

Table 6. Density of large wildlife trees killed by fire, but remaining standing 54-140 years later, in relation to previous stand age and topographic position within the MSdm1.

Previous stand age (years)	No. large wildlife trees/ha					
	Streamside			Upland		
	N	Mean	S.E.	N	Mean	S.E.
26-105	6	1.2	0.8	6	0.3	0.2
106-150	6	2.8	1.4	6	0.3	0.3
>150[1]	3	11.0	4.6	3	5.0	3.6

Effect	ANOVA test statistics		
	d.f.	F	P[2]
Log_{10} present stand age (as covariate)	1	0.42	0.52
Previous stand age (PSA)	2	5.39	**0.03**
Streamside-upland (SU)	1	9.51	**0.001**
PSA by SU interaction	2	1.10	0.35
Error	23		

[1] Mean of this class is significantly greater than both other classes.
[2] Significant P denoted in **bold**.

2.4.6 Coarse woody debris

There were no significant differences between streamside and upland stands in CWD volumes, number of logs counted per 90-m transect, or mean height of logs above ground in either the MSdm1 or ESSFxc (Table 7). Only in the MSdm1 was the log diameter significantly greater in the streamside stands than in the upland stands. Log size in these stands appeared to follow the same trend as live trees, i.e., live tree diameters were significantly larger in streamside MSdm1 stands than in upland MSdm1 stands. Because volume is a calculated function of both diameter (squared) and number of pieces counted, the coefficient of variation of volume (streamside 67%, upland 111%) was considerably higher than the coefficient of variation of diameter (streamside 26%, upland 32%). Therefore, the higher mean volumes of streamside MSdm1 stands could not be shown to be significantly different from upland MSdm1 stands, even though diameter was significantly higher and number of logs was slightly higher at streamside.

The CWD volume developed from the trees of the pre-fire stand was not a simple function of present stand age because it also appeared to depend on the size of the trees in the pre-fire stand. Based on the sizes of snags dating to the pre-fire stand, the sizes of logs on the ground from the pre-fire stand, or from veteran trees with burn scars, the age of the pre-fire stand could be estimated for most post-fire stands that were <150 years old. When pre-fire stands were <106 years old, virtually no CWD remained from that stand 86 years post-fire. The decay function of the logs from that pre-fire stand was impossible to estimate because no stands <86 years were found that also had a pre-fire stand age of <100 years. When pre-fire stands were 106-150 years old, high levels of CWD were present from that stand 54-63 years post-fire, but that CWD quickly decayed and much less remained 78-140 years

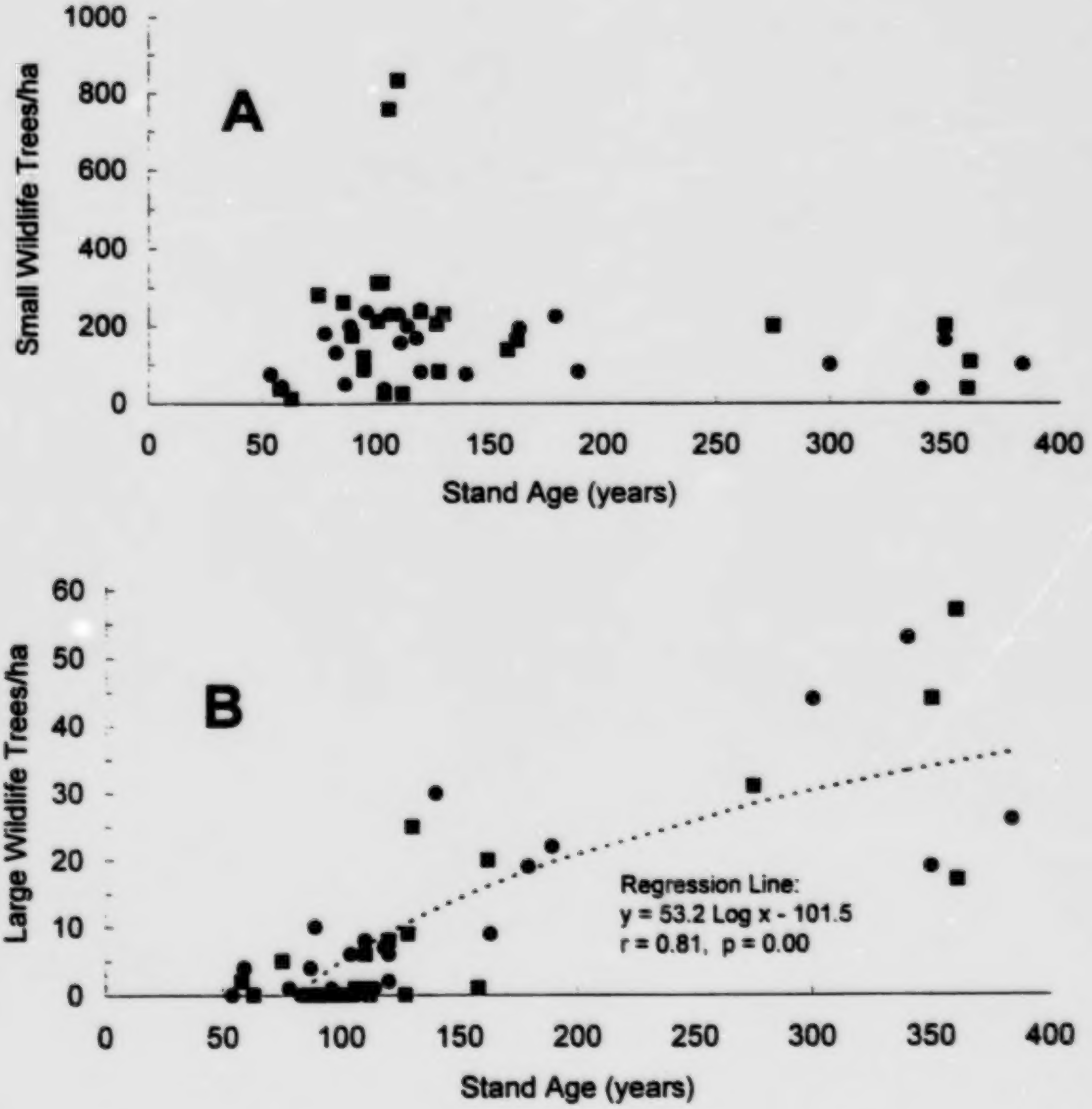


Figure 3. Density plotted against stand age for (A) <30 cm dbh and (B) ≥30 cm dbh wildlife trees developed from within post-fire stands for streamside and upland plots in the MSdm1 and ESSFxc.

Table 7. Paired streamside to upland comparison of CWD parameters in the MSdm1 and ESSFxc.

	MSdm1 (N=17 pairs)						ESSFxc (N=6 pairs)					
	Streamside		Upland				Streamside		Upland			
Parameter	Mean	S.E.	Mean	S.E.	t	P[1]	Mean	S.E.	Mean	S.E.	t	P
Volume CWD (m^3/ha)	169.1	27.6	117.2	31.8	1.84	0.08	115.8	44.6	136.0	42.7	1.09	0.32
No. logs/90 m	32.6	4.2	27.5	3.9	0.93	0.37	17.8	6.8	19.8	3.0	0.47	0.66
Diameter (cm)	17.4	1.1	14.4	1.1	3.55	**0.00**	20.0	5.2	19.3	2.1	0.12	0.91
Ht. above ground (cm)	13.0	2.3	9.8	1.4	1.08	0.30	3.6	1.7	4.7	2.0	0.62	0.56

[1] Significant P denoted in bold.

post-fire (except in a very decayed state, or class 5, and was consequently not counted in our surveys). The negative correlation with stand age was significant (r^2=0.54, $F_{1,11}$=13.09, P=0.004; Figure 4A) for the 106 to 150-year pre-fire stand age class.

When pre-fire stands were >150 years, CWD volumes remaining from the trees of the pre-fire stand were more variable, with some sites showing little or no remaining CWD, while others had high amounts. No correlation could be shown between stand age and CWD remaining from the trees of the pre-fire stand because the present stand ages (*x*-axis) covered a very narrow range from 96 to 110 years and was consequently not well suited to correlation measures. The wide variation in CWD levels remaining from these old pre-fire stands was probably dependent on species composition as well as on tree diameter. Both small and large lodgepole pine trees killed by fire appeared to rot at the base and fall within a few decades of a fire. Large spruce and subalpine fir trees appeared to remain standing longer after fire, and might therefore not be entering the decay cycle as quickly, so that CWD generated from that pre-fire stand might persist for well over 100 years after the fire.

The volume of CWD developed from trees growing within the post-fire stand increased significantly with stand age (r^2=0.74, $F_{1,44}$=126.25, P=0.000; Figure 4B). The mean was zero (i.e., no CWD developed from trees grown in the post-fire stand) at 75 years, increasing to 223 m^3/ha at 250 years, and 309 m^3/ha at 400 years. No significant differences could be shown using ANCOVA between streamside and upland stands, or between MSdm1 and ESSFxc stands.

Much higher CWD volumes than the predicted regression stood out in two cases (circled on Figure 4B). One of these points was a streamside site with a wide floodplain where a severe windstorm in October 1991 had toppled numerous large spruce. The other was a streamside site where a flash flood had released from a broken beaver dam two years previously and had undercut and toppled many streamside spruce. With the exception of these two points, most other points were fairly close to the regression line, and we may consider this line to be a fair approximation of the accumulation of CWD levels in the post-fire stand. It is not known if the upper maximum at 400 years is a steady state (i.e., as CWD decays into soil, it will be replaced by other trees falling), since there were no sites that could be dated as >400 years (since that is approximately the maximum tree age in the area).

The cumulative result of the two CWD development functions (Figures 4A and 4B) indicates that CWD will be at its lowest point approximately 100 years after fire (Figure 4C). By that time, most of the trees from the pre-fire stand have fallen and decayed to the point where they have been incorporated into the humus layer, and the trees from the post-fire stand would just be starting to contribute to the CWD layer. For stands that are <80 years when they burn, there will be very few large trees to carry into the next stand as CWD. These sites will enter the predicted CWD curve (Figure 4C) at a very low angle, with CWD always below 100 m^3/ha, and sometimes with almost no CWD at all until accumulation starts from the post-fire stand. When stands >150 years burn, the angle of the predicted CWD trajectory over time should fall off less steeply than for the 106 to 150-year-old stands, but the trajectory will be much more variable, depending on how long the dead trees remain standing. However, even the majority of these stands will have little CWD remaining from the trees of the pre-fire stand 100 years after a fire.

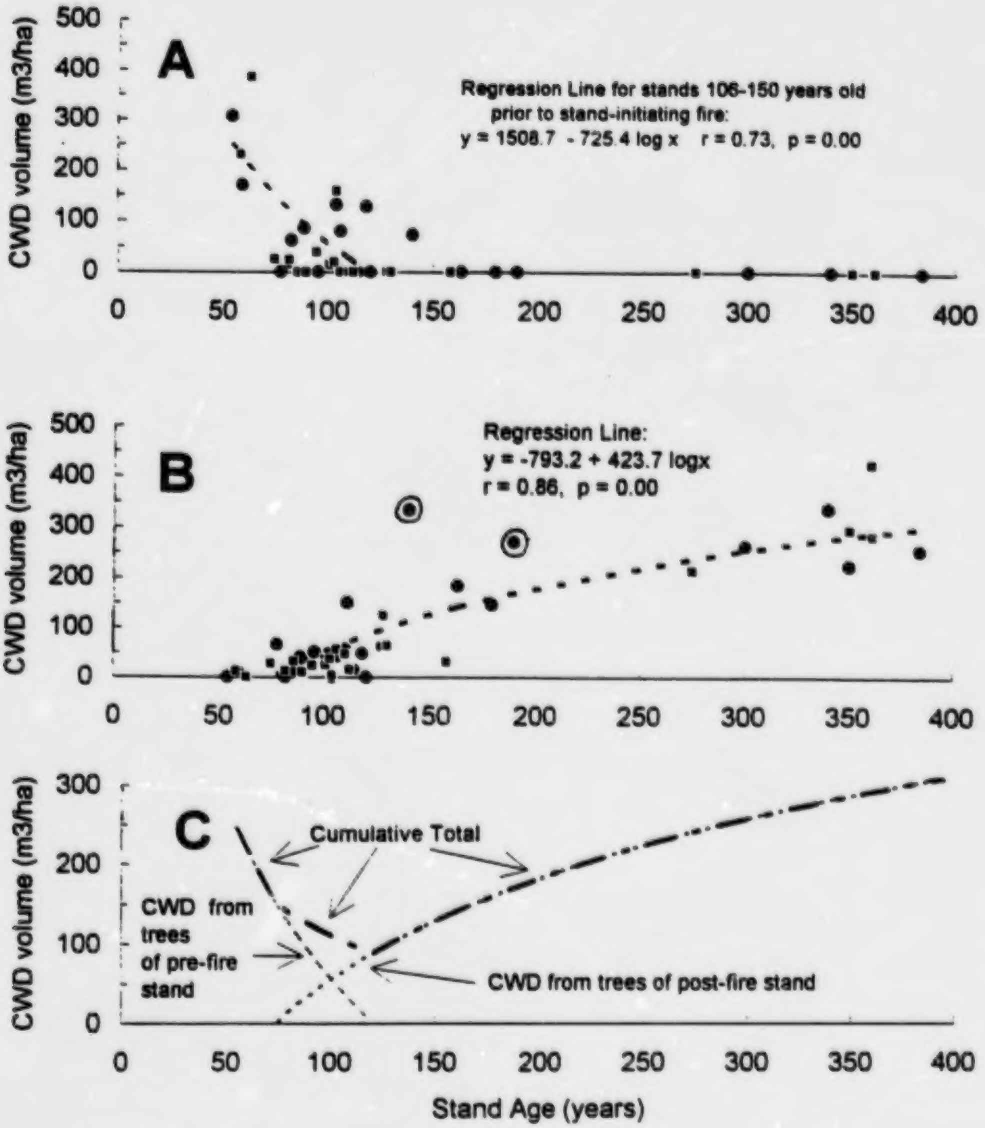


Figure 4. CWD volume developed from (A) the pre-fire, (B) the post-fire, and (C) both stands regressed against streamside and upland stand age.

2.5 Discussion

2.5.1 Natural successional paths

In the upland MSdm1, dense stands of lodgepole pine will establish after fire and will predominate for at least 200 years after fire with few exceptions. In streamside MSdm1 sites, spruce and lodgepole pine will generally establish in mixed species stands at densities lower than in upland sites. The predominance of spruce at very early stand ages made the streamside MSdm1 forests much more similar to ESSFxc forests than the upland MSdm1 forests. The riparian spruce vs. upland pine pattern is not repeated in the ESSFxc, where tree species diversity is high in both riparian and upland sites. By the time the forests reach an "old-growth" stage, the differences in tree composition between MSdm1 and ESSFxc forests, and between streamside and upland MSdm1 forests are very minor. Eventually almost all sites, in the absence of fire, would proceed to a mixed stand of subalpine fir and spruce. In the study area, this succession will be mostly complete by 250 years, but scattered lodgepole pine trees may persist until after 400 years.

The riparian spruce vs. upland pine pattern in the MSdm1 is therefore the result of several factors. Some riparian sites (estimated at 18% in the study area) will tend not to burn during large fires because of topography — deep gullies and wide, wet floodplains. Water bodies have often been cited as sometimes forming fire boundaries (Hunter 1993). In our case, there were riparian forests with wet soils rather than actual water bodies. On sites where gullies are not quite as deep, and floodplains not quite as wide, patches of trees or dispersed trees sometimes survived fires. On the remaining riparian sites, spruce tend to establish more successfully than lodgepole pine after fires, although most sites will have some lodgepole pine mixed in with the spruce.

The exact reasons for the establishment of spruce in riparian stands in preference to lodgepole pine are still difficult to pinpoint and were not directly examined here. Seed sources, germination success, and establishment success may all play a role. The riparian forests may, in general, be less dense than the upland forests because there are fewer good sites for germination in the wet, hummocky ground, or because germination may be poorer on the organic hummocks that are there. White and Engelmann spruce are often quoted as growing on rotting logs or stumps (see Harmon et al. 1986), although this is not a prerequisite for growth. In wet floodplain sites, the tree-growing sites were fairly limited to rotting logs and stumps, since this was the only ground elevated above the water table. Whether this alone could be the cause for spruce to be more common on floodplains than lodgepole pine was not addressed in this study, and would require microhabitat investigation of riparian tree-germination sites after fire disturbances.

Not all veteran trees survived the previous fires as canopy trees. Based on tree-ring sequences, many survived as suppressed seedlings or saplings that put on a growth spurt when the canopy around them was destroyed by fire. Within this study we did not measure tree-ring sequences of every veteran tree on our study plots. Only by such a fine analysis could the precise breakdown of the size ranges of trees that did survive fires be determined.

Successional paths after major disturbances other than fire, i.e., disturbances such as windthrow and insect outbreaks that do not necessarily lead to a new lodgepole pine cohort, will not likely alter the overall successional timing. However, they will alter the amounts of CWD and the densities of live trees and dead trees. These disturbances were not well studied here because, even though they may occur frequently, their effects will be overridden and thus hidden by the much more drastic effects of frequent stand-initiating fires in the area.

2.5.2 Recommendations based on natural disturbance regimes

Our understanding of whole ecosystems is very limited, so that accurately predicting all the effects of modern industrial forestry is very difficult. Consequently, one of the most common methods suggested to retain biodiversity in the face of industrial timber harvesting is to pattern timber harvesting as much as possible after natural disturbance regimes to which the biota has (presumably) already adapted (e.g., Swanson et al. 1993). If we apply the natural disturbance regimes that we detected here, as well as the natural successional paths of trees and the structures derived from trees (wildlife trees and CWD), we can assess the impacts of clearcutting and current riparian management guidelines and suggest possible methods of mitigation in riparian forests.

Forest Ecosystem Network Design — Within the MSdm1, the riparian corridors of spruce forest act as natural corridors for species that prefer spruce forest to the relatively undiverse upland lodgepole pine forests. Fires would more likely have left patches of forest along streams and beside wetlands and lakes rather than corridors of forest along streams. However, the riparian forests are acting like corridors of habitat in the landscape, even though they may mostly be the same age as the upland forests. This connectedness of spruce forest of high value to wildlife is not something that can be replaced by upland lodgepole pine forests. If the forests were all old-growth spruce forests, or all spruce-pine mixed forests as in the ESSF, then the value of these riparian forests in the MS would not be as high. Forest Ecosystem Network Design (at all scales from wide to narrow corridors) should favour riparian forest corridors to upland corridors in the MS. This may apply to all lodgepole pine-dominated forest systems in the western cordillera since the riparian spruce vs. upland lodgepole-pine pattern is common throughout.

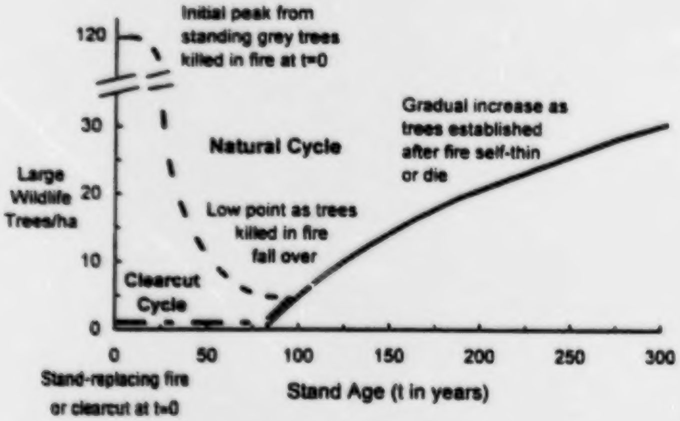
Live tree residuals after disturbances — Riparian timber-harvesting guidelines can be derived from the veteran tree data based on natural disturbances (see Table 4):

- 20% of the total stream length should have complete retention within at least 20 m of the stream bank in a given drainage or area (and these should be particularly located in deep gullies and at wide floodplains).
- 8% of the total stream length should have complete retention in small patches (<0.75 ha) at streamside, or within 20 m of the stream bank.
- 13% of the total stream length should have trees retained at a mean rate of 14 (range 10 to 18) for each 250-m stream reach harvested.
- 17% of the total stream length should have a few (1-5) trees retained for each 250-m stream reach.
- 42% of the stream length should have no trees reserved.

Similarly, we could apply our data to upland areas, even though that was not the intent of this project, and there may have been a clear bias because our upland sites were all within 100 m of streams. In that case, the recommended upland timber-harvesting guidelines would be:

- 73% of the forest base should be clearcut with no live tree retention.
- 4% should be retained in very small patches with complete retention.
- 10% should be partially harvested leaving about 10% of the trees (i.e., mean densities of 100 trees/ha left out of forests with mean densities of 1000 trees/ha).
- 13% of the land base should have 1-3 dispersed live trees/ha.

These recommendations address the retention of living trees after timber harvesting and consequently will only safeguard those species that rely solely on living trees (such as foliage-gleaning birds). This does not address the naturally very high levels of wildlife trees and CWD that would have been recruited from the forests of standing grey trees that usually remain after a fire. Consequently, these recommendations cannot be used literally with the expectation that biodiversity will be safeguarded unless the unnatural effects of dead tree removal during timber harvesting are also considered.



Note: Initial peak, Low point and the trajectory between them will depend on size and density of trees killed in fire.

Figure 5. Natural wildlife tree cycle in relation to stand age, and the projected cycle within managed stands after clearcutting.

Wildlife Trees — We can project our model of wildlife tree development in natural stands to the clearcuts and managed stands. Managed stands will be operating within a very small portion of the wildlife tree density vs. stand age plot (Figure 5). After clearcutting, no dead standing trees will remain because of Workers' Compensation Board (WCB) rules. The development of wildlife trees from any veteran trees remaining on site will be very slow (i.e., 87% estimated mean survival as living trees to over 100 years after fire). The occurrence of high levels of small wildlife trees may occur in the latter half of the 80-100 year projected rotation age, but only if self-thinning takes place. If juvenile stand spacing was undertaken, then very few of even these small wildlife trees may develop. These small wildlife trees are not used as much by wildlife compared to large wildlife trees, but overall numbers of small wildlife trees used are higher than large wildlife trees simply because there are more smaller trees, and the small trees appeared to be particularly important to bark-beetle specialist woodpeckers (see below). The high levels of large wildlife trees >30 cm dbh found in old-growth forests (20-60/ha) and after fires (where all trees die, this will be equivalent to the number standing alive before the fire) will not be duplicated at any time in a managed stand.

Cavity-nesting species are a very important component of the breeding bird community in the study area. They made up 22% of the 100 breeding bird species in the study area (see section 3) and 46% of the 26 permanent resident bird species. Cavity-nesting birds made up 8.7% of the 9614 birds detected during breeding bird sampling, even though this was probably an underestimate because most cavity-nesting birds are not well sampled by the song-bird counting methods used in the study. Six of the 22 cavity-nesting species were strong primary excavators (woodpeckers), while five were weaker excavators (chickadees and nuthatches). Eleven of the 22 cavity-nesting species were secondary cavity-nesting birds that depend on primary excavators to make nest holes that they can use. We simply do not know at this point the "minimum" wildlife tree density for a viable population of many of these species, or the minimum number of wildlife trees to maintain a population in an area — particularly for the strong primary excavators that feed on insects in wildlife trees as well as excavate nest holes used by themselves and other species. A considerable amount of research needs to be done before such predictions can be made with any degree of confidence.

If an entire landscape is clearcut, the consequent elimination of all wildlife trees will be disastrous to cavity-nesting wildlife. The most predictably successful mitigation will be retention of all trees — which are all potential wildlife tree recruits. However, this would halt all timber harvesting. Old-growth and immediate post-fire levels of large wildlife trees are probably not attainable in any managed stand, so that it will only be possible to meet these targets in places where timber harvesting will not take place. For the rest of the landscape, minimum targets should be those of natural stands around 100 years old.

These minimum targets for managed stands should be based on both species and densities. For example, large Douglas-fir in the area appeared to be the most highly used by woodpeckers and cavity nesters, and should be encouraged before all other tree species. Based on the data in Table 5, targets for wildlife tree retention during timber harvest in streamside MSdm1 riparian stands should be 1-3 large (>30 cm dbh) wildlife trees/ha in 80% of the stands, and 11/ha in the remaining 20% of the managed stands. These targets should be for dispersed wildlife trees, in addition to those that may occur in reserved patches, and are based on the levels of large wildlife trees killed by fire, but still standing 54-140 years later. Because WCB rules prohibit leaving standing snags on work sites, it is not entirely clear how this can be accomplished. Retention of 3-m high stumps (stubs) on a site may meet some of the wildlife tree needs, but we do not have data yet that will allow us to confidently set target levels. In particular, we need to know whether stubs can replace full-height trees (or how many stubs it may take) for cavity-nesting wildlife (that might need specific tree heights), and for birds that glean insects from bark or eat insects that live in decaying trees.

Managing for small wildlife trees (<30 cm dbh) may be even more problematic than for large wildlife trees because the small trees develop from within stands as the stands self-thin. In the natural young MSmd1 stands we examined, 12.8% of the mean of 1381 small standing stems/ha were dead. Of the total stems, 3.1% had been used by woodpeckers scaling bark to find bark beetles and 0.4% by woodpeckers for other foraging. Only one nest hole was found in these small wildlife trees (0.02% of the total small stems). Woodpeckers seeking bark beetles are therefore the primary wildlife users of small wildlife trees in young MSdm1 stands. These levels (3.1%) of bark-beetle attack are fairly low (compared to some of the much higher epidemic levels of Mountain Pine Beetle in the Okanagan Highlands that have triggered large-scale control and salvage timber harvesting) and are probably endemic levels. However, managing for stands that are susceptible to bark-beetle attack is antithetical to all present high-yield timber-management strategies. In particular, living trees in managed stands that have been thinned or spaced may never undergo self-thinning before the rotation age, and the trees may still be growing vigorously at the rotation age, so that they will not be susceptible to bark-beetle attack. If the bark-beetle specialist woodpeckers (i.e., Three-toed and Hairy woodpeckers) are to find a place in future managed forests, then this will have to be addressed.

Coarse Woody Debris — We can also project our model of CWD development in natural stands to clearcuts and managed stands (Figure 6). The high levels of CWD associated with old-growth stands and with young stands after fire will not be found in managed forests. While old-growth stands have always been recognized by the amounts of downed woody debris in the form of large logs, it is apparently only a fairly recent realization that young stands will also have high amounts of CWD (Harmon et al. 1986). The occurrence of high levels of CWD in young stands was substantiated in this study where the trees killed by fire are major contributors to CWD in the first 80 years after fire. Managed stands will be operating within a very restricted area of the naturally occurring CWD volume vs. stand age plot (Figure 6). Clearcutting and the maintenance of the often naturally high levels of CWD appear to be incompatible — you can have one or the other, but not both. It is therefore difficult to draw up any guidelines that could retain naturally occurring CWD levels.

Managed stands on first rotation after the cutting of original forest will have varying amounts of CWD, depending on how much was there to begin with in the form of downed logs and standing dead trees that were pushed over during clearcutting. Based on data collected in other parts of this study (see section 4), recent upland clearcuts in the study area average 59±11 m^3/ha (N=14). This CWD will likely decay in several decades, and for the latter half of the rotation (based on an 80 to 100-year rotation) there will be virtually no CWD remaining from the pre-clearcut stand. At 75 years after clearcutting, some CWD will start developing from within the stand — but only if self-thinning takes place. If stands were thinned as juveniles, then self-thinning may not take place and very little CWD may develop, even in the last part of the rotation. Managed stands on second rotation will have virtually no CWD after clearcutting, and the sites will thereafter be operating on a permanent lack of CWD.

In terms of biomass, the CWD might be partly made up for by fine woody debris (branches and needles left after timber harvest), but even then there is a projected long-term loss of nitrogen from a site (Harmon et al. 1986). In addition, some of the functions of CWD cannot be made up for by smaller woody debris. Logs decay more slowly and act as both water-storage sites and nitrogen-fixation sites (Harmon et al. 1986). In the wettest floodplains examined here, small woody debris would not provide the germination and tree-growing sites of fallen logs.

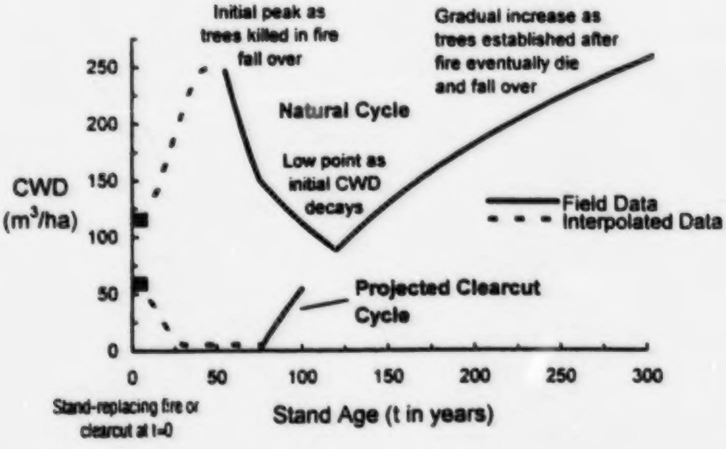


Figure 6. **Natural CWD cycle in relation to stand age, and the projected cycle within managed stands after clearcutting.**

Marten cannot use fine woody debris lying flat on the ground as winter den sites or as access points to hunting sites (Corn and Raphael 1992). Similarly, the Marten's principal prey, the Red-backed Vole, uses runways under large decaying logs in preference to other microsites (Hayes and Cross 1987; Tallmon and Mills 1994). One "cavity-nesting" bird, the Winter Wren, is actually a CWD-nester, nesting in large, upturned, rootballs (Harrison 1979), under logs (observed in this study), and even in logging debris piles near forest edges (Tobalske et al. 1991). Fine woody debris is unlikely to provide nesting opportunities for Winter Wrens.

Not all of the sites had high CWD levels, particularly those that had experienced two major fires in fairly quick succession (i.e., fires less than 80 years apart). It appeared that most CWD from an existing stand was consumed in a fire, and that CWD must then be developed from trees that were actually killed in the fire. When the trees of a young stand burn when they are small, then no CWD will be available for recruitment, and the little that there is will decompose fairly quickly. Walking through these sites was easy, even 80 or more years after a fire because there were very few logs on the ground. These naturally occurring sites are the closest mimics of the managed forests regenerating from clearcuts in terms of CWD. While these sites do exist, they are not common (3 of 17 sites in the MSdm1). The best guidelines that could be drawn up for CWD maintenance would then allow clearcutting over only 18% of the area (i.e., 3 of 17), and high-levels of retention would be recommended for the other 82% of the area, so that CWD could be recruited over time. The effects on wildlife of not applying these recommendations and largely eliminating CWD from most of an area are not entirely known (although we know it will be detrimental to some species as outlined in the previous paragraph). Examining the wildlife use of these sites in the MSdm1 that are naturally very low in CWD could help in predicting the ultimate effects on wildlife of reducing CWD levels.

2.5.3 Comparison of recommendations to the FPC guidelines

In this report, a landscape-level approach that will guide stand-level decisions with respect to live tree, wildlife tree, and CWD maintenance is advocated. The approach was based on naturally occurring levels in stands, and the proportions of those stands in the landscape. These recommendations based on naturally occurring conditions are very dissimilar to the uniform approach of the FPC riparian guidelines. In those guidelines, riparian buffers are based on stream width and stream "importance," i.e., fish-bearing streams and community watersheds are more important than other streams. This highlights the difficulties in establishing stand-level guidelines without taking note of the entire landscape. It is very difficult to recommend stand guidelines to be applied on a stand-by-stand basis when the guidelines are derived from landscape-level rather than stand-level patterns. Only by planning over larger areas can stand-by-stand decisions be justified in riparian management. Any uniform stand-by-stand approach will appear contrived when compared to natural disturbance regimes that show a lot of variability. This is not to suggest that riparian stand guidelines should be discarded because there is no "natural" equivalent. If that were the case, timber harvesting would be indefensible on the same grounds, since removing tree stems from a site has no natural equivalent.

The FPC riparian guidelines at present address only live-tree retention on riparian sites. On streams where there is a reserve zone (i.e., most of the streams in the MSdm1 because most streams are fish-bearing or in community watersheds), overall retention will be above and beyond the natural levels of live tree retention after fires. This is because fires often did cross streams and sometimes covered entire small drainages. However, these naturally occurring conditions do not present us with best-case scenarios for present management because fire-ravaged streams and drainages often undergo massive degradation through landslides and soil movements after every major fire (e.g., Meyer et al. 1992). On other streams, where there are no reserve zones, retention will be variable from a 5% minimum upward. The overall retention of live trees along streams should be similar, or higher, in absolute numbers to the natural regime, even though the dispersion of the trees will be different because retention will be based on stream width and our perception of stream importance rather than on topography and chance.

The dire predictions for the dearth of wildlife trees and CWD in the managed forests regenerating after clearcutting will not entirely apply to streamside and riparian forests, in which reserve zones are in place around streams and wetlands as recommended by the FPC riparian guidelines. Where complete reserve zones apply around all fish-bearing and community watershed streams, CWD and wildlife tree densities should remain high (whether these can "make up" for the lack of CWD and wildlife trees in adjacent uplands will not be addressed here). However, the "active floodplain" definition used is far too narrow and will not protect riparian forests that often extend more than 20 m from S3 streams, and there will not necessarily be a reserve on S4 streams <1.5 m wide. While Best Management Practices (BMP) recommend retaining all trees within 10 m of S4 streams, this will not apply where windthrow hazard is medium or high. Windthrow hazard is medium or high in most floodplain forests because of shallow rooting, so that the BMP reserve zone on S4 streams may rarely apply unless special considerations are made.

Stream channels <1.5 m wide were often observed to disappear entirely in low-gradient sections, with the water percolating through a wet, forested floodplain. Anybody measuring active floodplain width would not even find a channel to measure, and so not measure active floodplain width, when the real floodplain width was the entire valley bottom. The guidelines could be improved by defining riparian forest as the hygric to subhydric site series where they are in positions adjacent to streamflow or wetland. This would achieve the objective of special management for real floodplains, rather than hoping that most floodplains will be included in an arbitrary 10-m or 5-m distance from "active floodplain" boundary.

Where management zones are put in place that allow some timber harvesting, the results on CWD and wildlife tree maintenance are not easy to predict. The FPC riparian guidelines recommend 5% retention of mature trees along even the smallest of streams where other, more-stringent, recommendations do not apply to timber harvesting. This may address the need for retaining some living trees along many of the streams as found in the natural post-fire environment, but no matter how it is viewed, when 95% of the tree stems are taken off site, that is a net loss to the system in terms of the wildlife tree and CWD recruitment that would have taken place after a fire. The 5% of trees that do remain will continue to provide some large logs to create sediment traps and pools in small streams, but the long-term effects on wildlife and streams is not easy to predict. However, these sites will still be operating far below normal levels of CWD and wildlife trees.

To attempt to demonstrate the levels of CWD and wildlife trees at which riparian stands may be operating, assume an average "mature" streamside stand is between 80 and 140 years old when clearcut, and 5% of the trees will be retained after clearcutting. In an average MSdm1 stand, 121 trees are greater than 30 cm dbh, and there are only 433 total trees >20 cm dbh. If we consider half of the 765 trees in the 10 to 19.9 cm dbh class to be merchantable, then there may be about 800 "merchantable trees" per hectare. Leaving 5% implies that 40 trees will be retained per streamside hectare. Even if trees targeted for retention are the largest on site (i.e., all >30 cm dbh), still only 40 large living trees will be retained per hectare. Assuming 68 m^3/ha CWD will be retained on site from the previous forest after clearcutting (based on data from section 4), and that 13% will die naturally and fall over in a 100-year period, then CWD will still peak well below 100 m^3/ha in the 5% retention scenario, instead of at 200 m^3/ha or more as in the natural cycle. Wildlife tree levels will be 0/ha after timber harvest and will be recruited only very slowly (i.e., 5.2 over 100 years based on 13% mortality per 100 years) from the remnant trees on site. These levels can never equal those of old-growth forests, those of mature stands where the pre-fire stand had also been old growth, or any immature forest shortly after a fire has killed all the mature trees. The predicted levels may be similar, however, to levels in mature stands that grew up after fire had killed a very young stand. This also does not take into account the 140 small wildlife trees occurring per hectare. Some of these may be developed from within the managed stand, but only if self-thinning takes place rather than juvenile spacing at a point early in the rotation. Therefore, even given 5% retention of large trees, clearcutting of streamside forests will put them into a cycle that naturally occurs in only very few circumstances — and these naturally occurring circumstances are probably those least suitable for cavity-nesting birds.

Guidelines that retain more trees on site, or target a lower percentage removal from a site, will have quite varying results depending on actual conditions before harvest and the types of trees and numbers retained. Retention of higher levels of trees will, necessarily, result in higher levels of wildlife tree and CWD recruitment, and therefore put the sites closer to naturally occurring levels. We can assume that this will provide less risk to the high natural diversity of riparian zones. Just what levels of wildlife trees and CWD should be targeted in partially cut stands, and what the effects on wildlife will be, are still difficult to predict. However, because so many of the riparian sites are small headwater streams and lack fish, they will be clearcut with only 5% retention. Therefore, target levels of CWD and wildlife trees in partially cut sites should probably be closer to old-growth levels or those of a young forest recovering after a fire than the fairly low mature MSdm1 stand targets.

2.6 Conclusion

There are still large reserves of original forest in the forest base in the Okanagan Highlands because large-scale clearcutting only began in the early 1970s. (If there were not large reserves, this study would have been impossible because the unharvested study sites would not have been available.) The end results of clearcutting will result in managed forests that operate within a small subset of naturally occurring conditions of wildlife tree and CWD levels — and possibly within that subset that are least conducive to the retention of biodiversity. Much of the predicted lack of CWD and wildlife trees in riparian areas will be made up for by riparian management guidelines that retain reserve strips along important creeks and at least 5% of living trees on even the smallest creeks. However, it is doubtful that retaining these features only along riparian areas will fulfil the needs of wildlife that need these features in all forests. Only by recognizing how we are changing the natural cycles of tree-related features of the forests, and by attempting to better quantify what the effects on wildlife will be, can we design effective riparian management guidelines that indeed retain the uniqueness of these forests as well as their relationships to upland forests.

3 BREEDING BIRDS

3.1 Introduction

No published studies have specifically compared riparian breeding-bird abundance to upland breeding-bird abundance resulting from timber harvesting in the lodgepole pine forests of the western cordillera. Published studies in other forest types or habitats include McGarigal and McComb (1992), who found distinct riparian forest bird assemblages in Pacific Northwest forests; Stauffer and Best (1980) and Keller et al. (1993), who found bird species that were sensitive to the width of riparian forest corridors in an agricultural mosaic; and Croonquist and Brooks (1993) and Triquet et al. (1990), who found that preserving forest corridors along streams in clearcuts increased bird abundance and species richness relative to clearcutting to the stream edge in eastern forests.

The specific objectives of this breeding birds study in the Okanagan Highlands were to:

- Determine differences in breeding-bird composition and abundance between upland and riparian areas.
- Determine the effects of clearcutting, partial cutting, and reserve of corridors of forest on the breeding-bird composition and abundance of riparian areas.
- Make recommendations for timber-harvesting guidelines that would retain breeding-bird diversity in riparian areas.

3.2 Study Area

The study area was in the Okanagan Highlands east of Kelowna, B.C. (Figure 7). The northern part of the study area was in the Vernon Forest District and the southern part was in the Penticton Forest District, both in the Okanagan Timber Supply Area of the Kamloops Forest Region of the B.C. Ministry of Forests. The eastern part of the study area was in the Boundary Forest District of the Nelson Forest Region of the B.C. Ministry of Forests. The study area was chosen for two reasons. First, there were extensive existing clearcuts and a high priority to harvest much of the remaining forest because of the high susceptibility to Mountain Pine Beetle of the older lodgepole pine forests that dominate the area. Second, the area was a high plateau (1300-2000 m elevation) with numerous riparian sites including many low-gradient streams.

This study focused on the two predominant biogeoclimatic (BGC) zones of the Okanagan Highlands. The Okanagan Dry Mild variant of the Montane Spruce Zone (MSdm1) predominated at medium elevations (1300-1600 m), while the Very Dry Cold Engelmann Spruce–Subalpine Fir subzone (ESSFxc) and the Okanagan Dry Cold Engelmann Spruce–Subalpine Fir variant (ESSFdc1) of the Engelmann Spruce–Subalpine Fir Zone predominated at higher elevations (1600-2000 m) (Lloyd et al. 1990). The forest vegetation of the MSdm1 was dominated by a fire-caused disclimax of lodgepole pine in upland stands, which generally proceeds to a spruce (white-Engelmann hybrid) and subalpine fir climax in 400-500 years in the absence of stand-replacing fires. In fact, upland stands of spruce and subalpine fir were rare in the study area in the MSdm1. Spruce and subalpine fir were common at all stages of regeneration along creeks. Douglas-fir, western larch, and trembling aspen were occasionally found throughout the MS zone. The upland and riparian forests of the ESSF were dominated by Engelmann spruce and subalpine fir, although lodgepole pine was more abundant on sites until 300 or more years after stand-replacing fires. In this part of the report, results from the ESSFxc and ESSFdc1 were pooled and are collectively termed the ESSF.

The Kootenay Moist Cool variant of the Interior Cedar–Hemlock Zone (ICHmk1) was also common in the Okanagan Highlands at medium elevations. Sites from the ICHmk1 were climatically and floristically similar to the MSdm1. The ICHmk1 is transitional between the wetter ICH variants and the MS. It is dominated by lodgepole pine as is the MS, and lacks western hemlock, as does the MS. The BGC zones of the Okanagan Highlands have recently been remapped

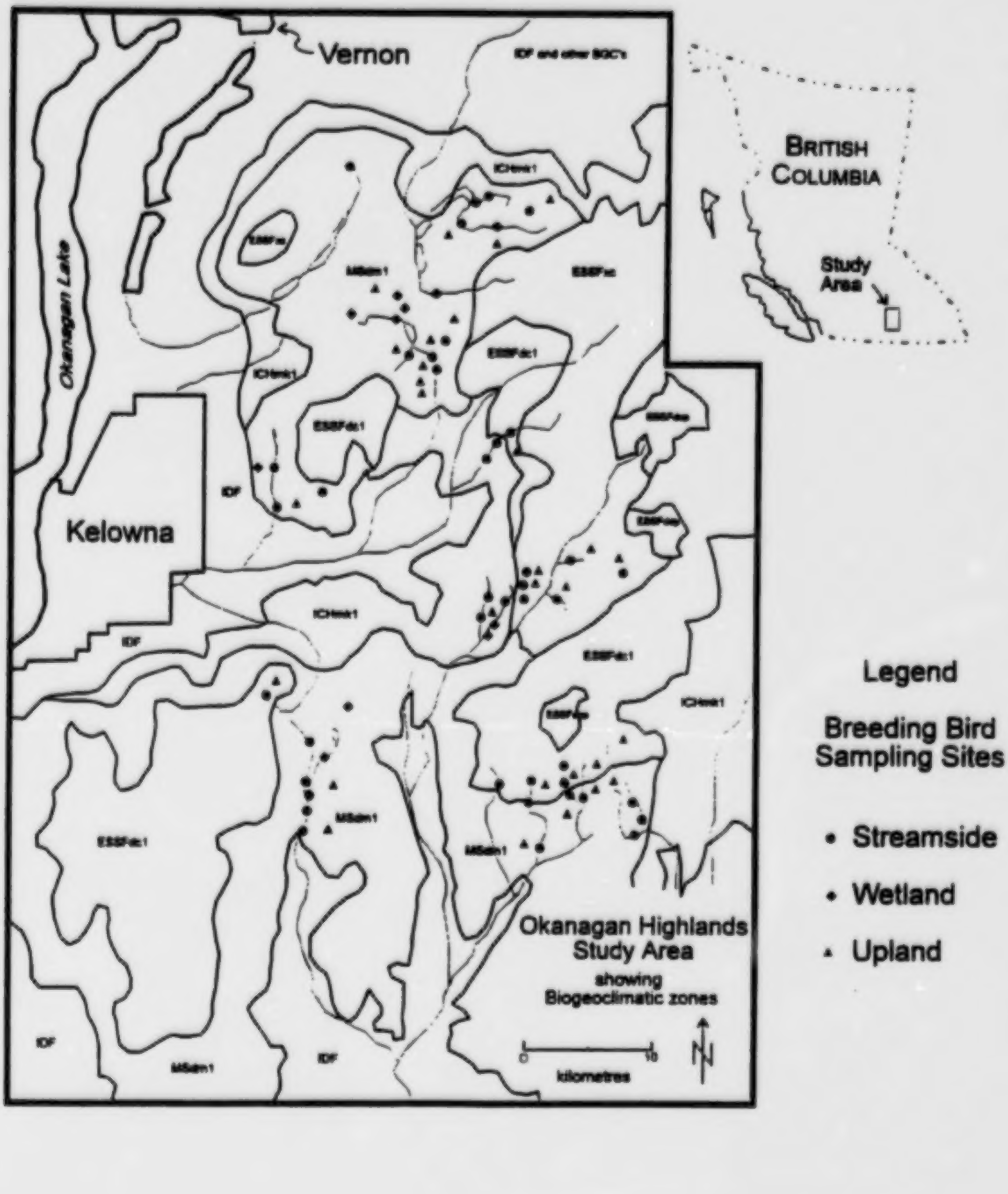

Vernon
IDF and other BGC's
ICHmk1
ESSFxc
MSdm1
ESSFdc1
Okanagan Lake
Kelowna
IDF
ESSFdc1
ICHmk1
MSdm1
Okanagan Highlands Study Area
showing
Biogeoclimatic zones
0
10
kilometres
N
BRITISH COLUMBIA
Study Area
Legend
Breeding Bird Sampling Sites
• Streamside
• Wetland
▲ Upland

Figure 7. Breeding bird study sites in the Okanagan Highlands.

at 1:250 000 (Braumandl and Curran 1992), and much of the land previously designated ICHmk1 has been redesignated as MSdm1. Classification of many particular spots often rely on site inspections — and even then, classification is not necessarily definite. Sites were used in two areas (West Kettle River and Boyce Creek) that were classified on the most recent BGC maps as ICHmk1. The West Kettle River sites were more similar to MSdm1 than to ICHmk1. The two Boyce Creek sites were closer to ICHmk1, with western redcedars being a subdominant tree. In the results and discussion, the ICHmk1 results have been pooled with the MSdm1 results.

In addition, several sites were used, which, although mapped at 1:250 000 as MSdm1, were near the boundary of the ESSFdc1, and were actually more similar floristically to the ESSFdc1 than MSdm1. For this analysis, these sites have been considered ESSF sites.

3.3 Methods

3.3.1 Study design and study sites

This was a retrospective study that took advantage of existing sites with various cutting regimes adjacent to streams or wetlands. The range of sites that could be examined within this study was therefore dictated by the availability of streamside and upland sites where timber harvesting had already taken place.

Two separate investigations were undertaken: densities of 33 common bird species were compared between streamsides and uplands in the MS and ESSF under three timber-harvest treatments (control forest, corridors of forest, and clearcutting) using 60 plots; densities of 20 common bird species in wetlands compared to wetland borders in the MS under three timber-harvest treatments in the borders (control forest, corridor of forest, and clearcutting) using nine plots. Additional analyses were made to determine the effects of forested corridor width, the effects of level of partial timber harvest in streamside forest corridors, and the effects of clearcutting to one side of a stream on bird communities, using 15 plots in addition to those sampled in the previous two investigations.

3.3.2 MS and ESSF streamside riparian investigation

Experimental design was a three-factor analysis of variance (ANOVA) using 60 plots. The first factor was BGC zone with two levels — MS and ESSF, as discussed in section 3.2. The second factor was topographic position with two levels — streamside riparian and upland. The third factor was timber-harvest treatment with three levels selected: control (or unharvested), reserve of a corridor of forest, and clearcut. The ANOVA study design had a total of 12 cells (2 BGC zones × 2 topographic positions × 3 timber-harvest treatments) with five replicates selected per cell. When individual cells were referred to, the specific level of each of the three factors was used to name the cell. For example, MS Riparian Control refers to a specific cell with five replicates. When specific levels of a factor or specific combinations of levels of several factors were referred to that averaged over other non-significant factors, only the levels of the factor(s) under consideration were used to name the effect referred to. For example, MS Riparian refers to the average of all levels of timber-harvest treatments that were within the MS Riparian.

All streams chosen for this study were picked first from 1:15 000 air photos. All streams had distinct banks with bank-to-bank widths between 0.6 m and 7 m. Extent of riparian floodplain forest was found to be unrelated to stream width over this range of widths (see section 2).

To reduce the potential habitat differences and the potential for variability in bird communities, streams were selected that did not have open wetlands, beaver ponds, or beaver marshes in the sampled stream reach, and with floodplains that were bounded by gully walls less than 15 m tall. The gully wall width could often only be measured when actually in the field, so that some of the sites originally chosen were later rejected on this basis. Streams in deep gullies were also usually too noisy to hear birds singing or calling effectively. Wherever possible, streams with gentle gradients were chosen in preference to those with steep gradients to increase the probabilities of successfully hearing the birds that were singing or calling during sampling.

Upland sites selected were at least 100 m from streams to avoid riparian effects. However, small (<0.1 ha) pothole riparian areas or old glacial meltwater channels were abundant in the upland because of the rolling nature of the post-glacial landscape on the plateau. Upland plots were chosen from 1:15 000 air photos prior to entering the field, with every effort made to choose upland plots that were uniform upland habitat. However, some small wetlands and ephemeral streams were inadvertently included on the chosen plots because they were too small to be identified easily on air photos. These were found to add some variability to the upland bird species composition. In particular, Northern Waterthrushes and Townsend's Warblers found in upland MS habitats were almost always associated with these small features. (Scientific names of birds are listed in Appendix I.)

Control sites were placed in continuous forest or forested corridors of at least 500-m width, since preliminary sampling had indicated that breeding-bird abundance in corridors wider than 500 m could not be shown to be different from continuous forest. Streamside forest corridors that had forest reserves on both sides of the stream, and were between 31 and 163 m in total width, were selected for this investigation. Upland forest corridors used for comparison were between 74 and 188 m in total width. Virtually all merchantable trees had been removed from clearcut sites. Along streams, more of the non-merchantable understorey trees and shrubs tended to be retained after clearcutting than on the upland sites.

3.3.3 MS wetland riparian investigation

Experimental design was a two-factor ANOVA using nine plots. The first factor was topographic position with two levels — wetland or border of the wetland. This border was not termed "upland" because it often contained both upland and riparian forest, or at least had contained both before clearcutting. Riparian forest was defined here as areas where trees were growing primarily on organic hummocks with standing water between hummocks. The second factor was timber-harvest treatment in the border — control, corridor, or clearcut, as above for streamside sites. Since the timber-harvest treatment was actually applied only to the border area, the interpretation of significant timber-harvest treatment effects will indicate direct results in the border area, but only indirect results when applied to the wetland area. Therefore, within the ANOVA results, the wetlands that were bordered by a corridor of forest were referred to as "corridored wetlands" to imply that the treatment was not on the wetland itself.

Wetlands investigated were at least 3 ha in size. All wetlands were used by beaver and all but one were at least partially ponded by beaver. The exception was the upper end of a large reservoir that was filled to its highest level throughout the sampling period, and in which shrub and emergent sedge cover were similar to other sites. Vegetation cover levels and composition varied both between and within sites. Willow and alder shrub wetlands with standing water were the most common wetland type, followed by areas of emergent sedges, and by open ponded water. Conifers (mainly spruce and subalpine fir) were sparsely distributed in some of the wetlands. Standing dead trees from fairly recent flooding were only abundant at one of the sites.

3.3.4 Corridor-width and partial timber-harvest analyses

The effects of forest corridor width were examined using samples from forests adjacent to streamsides and wetlands from the first two investigations (see above); these were supplemented by seven additional streamside forest corridors of a variety of widths in the MS. A one-factor ANOVA was used to compare bird abundance between corridor width classes.

The effects of partial timber harvesting were examined in six partially harvested additional streamside corridors as well as two partially harvested upland corridors, all in the MS Zone. One of the partially harvested streamside sites was actually the site of severe windthrow (48-82%) rather than timber harvest. The effect of opening the canopy was similar to timber harvesting, although the downed trees were not salvaged because they were small-diameter lodgepole pine and of minimal commercial value. A one-factor ANOVA was used to compare bird abundance between partial harvesting levels and total reserves. Corridor-width effects were controlled by comparing partially harvested corridors to the same number of replicates of corridors of comparable widths.

3.3.5 Sampling methodology

Five-hectare plots, 500 m long and 100 m wide, were established at all sites used in the first two investigations (Figure 8). For streamside riparian sites, plots were centred along a 500-m stream reach with plot boundaries 50 m to either side of the stream banks. The 50-m width from the stream bank included varying amounts of valley wall and upland terraces, depending on the actual width of floodplain or valley bottom. For upland sites, plot boundaries were 50 m to either side of the centreline of a forested corridor, or of a straight line at control or clearcut sites. For wetland riparian sites, plots were centred on a 500-m length of the wetland/forest boundary. Where sites were clearcut to wetland edge, the edge of the clearcut was made the centreline of the plot.

All 5-ha plots were divided into four 1.25-ha subplots to facilitate recording bird detections. Birds were recorded separately for each of these subplots as the observer walked the plot. For 5-ha streamside corridor, wetland border, or upland corridor plots that were very different in corridor width or level of partial cutting from one end of the plot to the other, using subplots allowed data to be tallied for subplots that were more homogeneous. In addition, two partially harvested streamside sites were selected for study that were only 250 m long, and this allowed them to be included in the comparisons.

The limiting factor in plot width was the high noise level of some of the streams. To increase the chance of detecting all birds singing or active on a plot, the subplots were sampled separately, first by walking one side of the stream at a rate of 1-2 km/hr and noting all birds on that side of the stream, then crossing the stream and sampling the other side at the same rate in the reverse direction. The easiest route of travel was used and was usually ±20 m from the stream bank. Only by using such a narrow search range could all singing birds on a plot be heard along noisy streams, and their locations judged with sufficient accuracy to determine if they were on the side of the plot being sampled. Some birds certainly crossed the stream between the time that subplots were sampled. To equalize any possible bias, upland sites were sampled in the same fashion as the streamside plots, with the plot centreline used as a reference line instead of the stream.

Both sides of the wetland plots were sampled by an observer walking once along the wetland/forest boundary because there was no stream noise to cause interference.

Breeding birds were sampled between 20 May and 1 July 1994, and 22 May and 29 June 1995. Each site was sampled six times at approximately weekly intervals. Individual plots were sampled only in one of the years. One observer (the author) sampled plots in both years, while different second observers were used in 1994 and 1995.

Sampling took place between a half hour before sunrise (0415-0430 hrs) and 1000 hrs PDT. Between three and seven 5-ha plots could be sampled per day by one observer, depending on the distances to walk to plots from roads, and the road distances between plots. Time-of-day of sampling at each site was rotated, with each site sampled at least once at sunrise. Whenever possible, weekly sampling of plots was alternated between the two observers. Direction of sampling each plot was also alternated: when an observer resampled a plot, it was done in the opposite direction from the last time that the observer had sampled the site. All birds seen or heard in the plot were recorded. Information collected included species, number in a group or flock, behaviour (singing, other territorial behaviour, flying over the site, or other), topographic position (floodplain, slope, or upland), and habitat (forest, clearcut, or partial cut).

Topography, forest boundaries, and other landmark distances were measured using tape or hip chain, and sketch maps were produced of each site to assist observers in distance estimation while bird sampling. Beginning and end points of each 1.25-ha subplot were flagged in the field.

Since forest corridors and partially harvested sites were not logged specifically for this study, corridor width and level of partial harvest had to be measured. Exact forest corridor widths were measured at three or more points along the 500-m length of each of the plots. Levels of partial harvesting were determined by tallying trees (>10 cm dbh) and stumps (>10 cm estimated dbh) in eight 7-m radius subplots at 50-m intervals in all plots where partial cutting took place. Level of partial cutting was expressed as the percentage of trees removed from a site [(no. stumps ÷ no. trees + stumps) × 100].

3.3.6 Data analyses

In general, bird-community macroparameters do not yield significant insights into community processes, except for sometimes highlighting species groups deserving further study (Wiens 1989). Consequently, their consideration was not emphasized here. Only total abundance of birds was examined statistically within the ANOVA study designs.

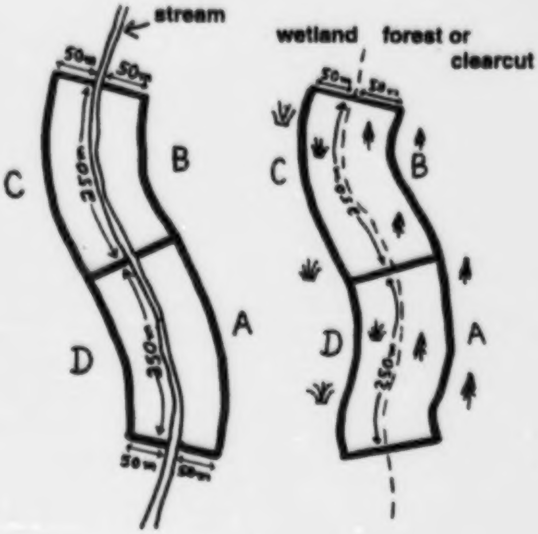


Figure 8. Typical layout of subplots within breeding-bird sampling plots of 5 ha in wetland and streamside situations.

For the first two investigations, where constant-size, 5-ha plots were used, direct comparisons were made of the number of birds detected. Total abundance was analyzed first, followed by any species with an overall mean abundance of >1.0 per cell (considering only levels of factors where present) or by any species with a coefficient of variation of <150%. General Linearized Model (GLM) ANOVA techniques were used throughout.

3.3.7 MS and ESSF streamside riparian investigation

For those species where significant three-way interaction effects were found, the study design effectively devolved into a one-way ANOVA that compared the 12-cell means. This only occurred for three species and for total abundance, and, in each case, most of the two-way interactions within levels of the other factors were also significant, so that cell means would have been compared in most cases at any rate. Where no significant three-way interaction effect was found, the three possible two-way interaction effects were examined. For factors not involved in any interaction effect, main effects were examined for significance, and if significant, then marginal means were compared. For those factors involved in significant two-way interactions, row and marginal means within each of the factor levels were compared, averaging over the other non-involved factor. Probability of a Type I error (α) was set at 0.05. Power ($1-\beta$) was calculated for each comparison made.

The sampling took place over two field seasons, although individual plots were sampled in only one of the field seasons (i.e., either 1994 or 1995). The possible confounding effects of inter-year population fluctuations therefore needed to be examined. In 8 of the 12 cells, three of the five replicate plots per cell were sampled in one of the years, with the other two replicate plots sampled in the other year. The species densities for the 40 plots of these 8 cells were examined for annual differences. Half of these 40 plots were sampled in 1994, and half in 1995.

3.3.8 MS wetland riparian investigation

Cell means were compared within levels of the factors when the two-way interaction effect was significant. The probability of a Type I error was reduced accordingly for the number of within-factor comparisons that were made. Power of the analyses was calculated for all interaction and main effects. In the absence of a significant two-way interaction, row and column marginal means were tested for significance. Probability of a Type I error (α) was set at 0.05. Power ($1-\beta$) was calculated for each interaction and simple main effect analyzed. All sampling was carried out in 1995 so that no inter-year effects needed to be examined.

3.3.9 Corridor-width and partial timber-harvest investigation

One-factor ANOVA was used to compare detection densities of species groups between forested corridor-width classes. The classes were not predefined, but were made after the actual forest-corridor widths had been measured in the field. Initial classifications attempted were by 25-m groups (i.e., 0-25 m, 26-50 m, etc.), but these were modified slightly so that each class contained at least four replicates, and the final width classes (see Table 11, section 3.4.6) reflect the actual corridor widths within each class. For forest corridors that differed greatly in width or level of partial harvest within single 500-m-length plots, the 500-m plots were divided into two 250-m plots, and these were treated as independent samples.

After examination of the breeding-bird results from the first two investigations, MS upland corridors, where lodgepole pine predominated, were treated separately from all other corridors where spruce and subalpine fir were more common. One-factor ANOVA was used to compare bird abundance within classes of partial harvesting levels, controlling for corridor-width effects by only using comparable corridor widths. An additional one-factor ANOVA was made for the streamside corridors that existed on only one side of the stream (clearcut to the other side) compared to similar-width streamside forest corridors that spanned both sides of the stream. These samples were drawn from within the total corridors previously sampled. Probability of a Type I error (α) for all analyses was set at 0.05. Power of the analyses ($1-\beta$) were only calculated effects where numbers of replicates per treatment were equal. Duncan's Multiple Range Test was used to determine differences in width-class means when the main ANOVA effect was significant.

Detection densities were used for these analyses because in narrower corridors or wetland borders the forests often occupied only a small portion of the 5-ha plot. Number of birds detected in forest habitat were divided by the forested area and then divided by six, since each site was sampled six times. This yielded a density of detections per forested hectare.

Species-by-species analyses within this investigation generally had very low power because there were often less than five replicate samples within any sub-group of corridor-width classes or partial-harvesting levels. Therefore, bird species were grouped according to their responses to timber harvesting and use of forest habitats (see list in Appendix G). Results from the first two investigations of this study were used to group common species, while uncommon species were grouped based on the little data that were collected, personal familiarity with the habitat associations in the area, or on general literature sources (e.g., Godfrey 1986; Cannings et al. 1987).

Species groups used for this analysis were forest species, open-forest species, generalist species, and other species. The "forest species" group was subdivided into "width-sensitive species" and "other forest species." Width-sensitive species were those common species whose density declined in forest corridors <72 m in total width. Where sample size permitted, "common other forest species" were separated from "uncommon species" to refine the analysis. "Open-forest species" consisted of those that were most abundant in forest corridors rather than closed forest, or could tolerate a considerable amount of canopy opening in partially harvested stands. "Generalist species" consisted of only two species — Dark-eyed Junco and Pine Siskin — that were abundant in many habitats. "Other species" were those more typical of clearcuts or wetlands.

3.4 Results

3.4.1 MS and ESSF streamside riparian investigation

Within the 60 5-ha plots in this portion of the study, 5896 birds were detected. Analyses were completed individually for 33 species of birds; an additional 41 species were considered uncommon and could not be analyzed separately (see Appendix D). Year-to-year differences in abundance were only significant for four species. Brown Creepers and Red Crossbills were more abundant in 1994, and Evening Grosbeaks and Pine Siskins were more abundant in 1995. Interpretations made of the results for these four species (Table 8) took into account any possible confounding effects of these annual fluctuations.

Total bird abundance was significantly affected by the factors under study, but there was no simple relationship with any one of the factors (Table 8). In general, abundance was highest in MS riparian corridors and clearcuts, ESSF upland control forests and corridors, and ESSF riparian clearcuts. Abundance was lowest in MS riparian control forests, MS upland control and corridor forests, and ESSF upland clearcuts. Other treatments were intermediate in bird abundance. Species richness was not analyzed separately because it was significantly correlated with total abundance (r^2=0.72; P=0.000; N=60, see Appendix A). There did not appear to be any levels of the factors (or combination of levels) that deviated from this correlation.

3.4.2 Effects of timber-harvest treatment

Of the 33 common species for which analyses were completed, 30 species showed some significant effects due to timber-harvest treatment (Table 8). Within these 30 species, four basic groups could be identified. The first group included 11 species significantly most abundant in forests. The second group included three species significantly most abundant in forest corridors. The third group included five species that were equally abundant in forest corridors and in clearcuts ("edge" species). The fourth group consisted of 11 species most abundant in, or limited to, clearcuts.

The 11 species of the first group were significantly most abundant in forested habitat: Boreal Chickadee, Brown Creeper, Evening Grosbeak, Golden-crowned Kinglet, Hammond's Flycatcher, Red-breasted Nuthatch, Red Crossbill, Swainson's Thrush, Townsend's Warbler, Varied Thrush, and Winter Wren. Of this group, only Evening Grosbeaks were ever commonly found in clearcuts, but in almost all cases they were simply flying over the plot from one stand to another.

The three species of the second group were significantly more abundant in the forest corridors of this investigation than in completely unharvested forests or in clearcuts: Hermit Thrush, Ruby-crowned Kinglet, and Yellow-rumped Warbler. The corridors were not partially harvested, although each of these species could also be common in partially harvested corridors (see 3.3.4). The Hermit Thrush was included with this group, although the association was not as strong as for the other two species. They were also common in some of the clearcuts that had regenerated with conifers to >2 m in height, and this obscured the strength of the association with corridors. Both Ruby-crowned Kinglets and Yellow-rumped Warblers become abundant in regenerating forests that are approaching 7 m in height (L. Gyug, pers. obs.). Consequently, each of these three "corridor" species may be better thought of as "young-forest" species that will also use forest corridors extensively.

The five species of the third group were equally abundant in forest corridors and in clearcuts, and might be termed "edge" species: Brown-headed

Table 8. **Significant factor effects for common bird species and total abundance of birds in the MS-ESSF streamside riparian investigation.** Full data is reported in Appendix A, and full ANOVA statistics in Appendix B. Scientific names of species are listed in Appendix I.

COMMUNITY PARAMETERS	C.V.[1]	Treatments (Mean no. detected in 5-ha plots visited six times each)
Total abundance	30%	Not significant 3-way interaction effect (P=0.053), but since all 2-way interactions also significant, 1-way ANOVA used; Four groups of not significantly different treatments (in descending order): 1. MS riparian corridor = MS riparian clearcut = ESSF upland corridor = ESSF upland control = ESSF riparian clearcut 2. MS riparian clearcut = ESSF upland corridor = ESSF upland control = ESSF riparian clearcut = ESSF riparian corridor = ESSF riparian control 3. ESSF upland control = ESSF riparian clearcut = ESSF riparian corridor = ESSF riparian control = MS upland clearcut 4. ESSF riparian corridor = ESSF riparian control = MS upland clearcut = ESSF upland clearcut = MS upland corridor = MS riparian control = MS upland control
SPECIES	**Total # birds**	
American Robin	115	Clearcut (5.40) > Corridor (2.10) and Control (0.15)
Boreal Chickadee	148	Only in ESSF; Absent from clearcut; No other significant effects
Brown-headed Cowbird	146	Absent from control forest Riparian corridor and clearcut (3.75) > Upland corridor and clearcut (0.65)
Brown Creeper	130	Absent in clearcut; No other significant effects (based primarily on 1994 results, when they were relatively abundant)
Chipping Sparrow	124	Clearcut (4.15) > Corridor (0.95) and Control (0.80)
Dark-eyed Junco	50	ESSF control (5.4) > MS control (3.4) MS corridor (16.7) > ESSF corridor (7.2) Riparian corridor (16.2) > Upland corridor (7.7) Within MS: Clearcut (19.6) and Corridor (16.7) both > Control (3.4) Within ESSF: Clearcut (14.0) > Control (5.4), but Corridor (7.2) not different Within riparian: Corridor (16.2) and Clearcut (16.0) both > Control (4.5) Within upland: Clearcut (17.6) > both Corridor (7.7) and Control (4.3)
Dusky Flycatcher	168	Absent in control forest; Upland clearcut (3.6) > Upland corridor (0)
Evening Grosbeak	140	ESSF control (10.9) > MS control (1.5); ESSF clearcut (6.2) > MS clearcut (0.1) Within MS: Corridor (2.1) > Clearcut (0.1) but Control (1.5) not different from either (based primarily on 1995 results, when they were relatively abundant)
Golden-crowned Kinglet	66	Virtually absent from clearcuts (1 of 567 detections), so not included in analysis MS riparian (16.7) and ESSF Upland (19.3) > MS Upland (8.1) ESSF upland (19.3) > ESSF Riparian (12.5)
Gray Jay	144	No significant effects, although absent from MS clearcuts
Hammond's Flycatcher	197	Absent in clearcut; Absent in upland; No other significant effects (very high C.V.)
Hermit Thrush	105	ESSF (3.23) > MS (1.47) Riparian (3.00) > Upland (1.70) Corridor (3.15) > Clearcut (1.55), but Control (2.35) not different from either
Lincoln's Sparrow	64	Absent in control forest MS clearcut (10.7) and ESSF corridor (1.1) > MS corridor (0) Riparian clearcut (14.4) > Upland clearcut (2.3) and Riparian corridor (1.1) Upland clearcut (2.3) and Riparian corridor (1.1) > Upland corridor (0)

[1] Coefficient of variation = Mean of (cell standard deviation / cell mean) × 100, for all cells in which species were detected.

Table 8 (continued)

SPECIES	Total # birds	Treatments (Mean no. detected in 5-ha plots visited six times each)
MacGillivray's Warbler	129	Significant 3-way interaction effect; from 1-way ANOVA, 2 groups of not significantly different treatments (in descending order): 1. MS riparian clearcut = MS riparian corridor = ESSF riparian clearcut = ESSF upland corridor = MS upland clearcut 2. ESSF upland corridor = MS upland clearcut = All other treatments
Mountain Bluebird	161	Only found in clearcut; No other significant effects
Northern Flicker	148	Absent from control; Clearcut (0.90) > Corridor (0.25)
Northern Waterthrush	173	No significant differences (very high Coeff. of Var.)
Orange-crowned Warbler	154	MS (2.4) > ESSF (0.9) Clearcut (3.40) > Corridor (1.40) and Control (0.15)
Pine Siskin	74	Overall: ESSF (12.37) > MS (6.70) Within control sites: 1995 (14.9) > 1994 (7.5); ESSF (15.4) > MS (7.0); Upland (14.3) > Riparian (8.2)
Red-breasted Nuthatch	133	Significant 3-way interaction effect; From one-way ANOVA, 3 groups of not significantly different treatments (in descending order): 1. ESSF upland corridor = ESSF upland control 2. ESSF upland control = MS riparian corridor 3. All other treatments equal
Ruby-crowned Kinglet	133	Significant 3-way interaction effect; From one-way ANOVA, 4 groups of not significantly different treatments (in descending order): 1. ESSF upland corridor 2. ESSF riparian corridor = MS riparian corridor 3. MS riparian corridor = All other treatments except MS upland clearcut 4. All other treatments equal including MS upland clearcut
Red Crossbill	136	Upland (2.43) > Riparian (0.33) Control (2.65) > Clearcut (0.15) but Corridor (1.35) not different from either (based primarily on 1994 results, when they were relatively abundant)
Savannah Sparrow	163	Only found in clearcut; No other significant effects
Song Sparrow	150	Absent from upland; Absent from control forest; Clearcut (1.6) > Corridor (0.1)
Swainson's Thrush	95	Corridor (3.40) and Control (2.45) both > Clearcut (0.25)
Townsend's Warbler	77	Absent from clearcut; MS riparian (18.6) & ESSF upland(15.3) > MS upland (2.0)
Varied Thrush	140	Absent from clearcut; ESSF (2.4) > MS (0.35)
Warbling Vireo	163	Absent from control; No other significant effects
White-crowned Sparrow	110	Absent from control; Clearcut (7.40) > Corridor (0.65)
Willow Flycatcher	140	Absent from control and Corridor; Riparian (3.8) ≥ Upland (0.2) (P=0.052)
Wilson's Warbler	148	Riparian (7.13) > Upland (2.03) Corridor (7.10) and Clearcut (5.80) both > Control (0.85)
Winter Wren	114	Riparian control (5.8) > Upland control (0.8) Riparian corridor (4.8) > Upland corridor (0.4) Within riparian: Control (5.8) and Corridor (4.8) both > Clearcut (0.3)
Yellow-rumped Warbler	59	MS corridor (17.0) > ESSF corridor (11.7) Within MS: Corridor (17.0) > Control (7.0) > Clearcut (1.8) Within ESSF: Corridor (11.7) > both Control (6.2) and Clearcut (3.3)
		Upland control (8.5) > Riparian control (4.7) Riparian clearcut (4.1) > Upland clearcut (0.8) Within riparian: Corridor (13.9) > both Control (4.70) and Clearcut (4.1) Within upland: Corridor (14.8) > both Control (8.5) and Clearcut (0.8)

Cowbird, Dark-eyed Junco, MacGillivray's Warbler, Warbling Vireo, and Wilson's Warbler.

The 11 species of the fourth group were most abundant in, or limited to, clearcuts: American Robin, Chipping Sparrow, Dusky Flycatcher, Lincoln's Sparrow, Mountain Bluebird, Northern Flicker, Orange-crowned Warbler, Savannah Sparrow, Song Sparrow, White-crowned Sparrow, and Willow Flycatcher.

The three common species not significantly affected by timber-harvest treatment did, in fact, appear to be affected, but the sampling regime was likely inadequate to detect the effect. Even though Gray Jays were absent from MS clearcuts, no significant timber-harvesting effect could be shown, since they appeared to be using a wide variety of habitats and were not as commonly detected on individual plots as territorial songbirds. They had finished breeding by the census period and the family groups appeared to be wandering widely. Pine Siskins require trees for nesting, but are not territorial, using large breeding territories covering a variety of habitats. For both of these species, 5-ha plots designed for territorial songbirds were probably not the best sampling methods. The Northern Waterthrush appeared to be more responsive to microhabitats within plots than to the factors under study at the 5-ha plot level. This resulted in a high coefficient of variation that obscured possible responses to timber-harvesting treatments.

3.4.3 Effects of topographic position

Of the 33 common species analyzed, 14 showed some significant association with streamside habitats and two with upland habitats (Table 8). For most of these species, the associations with topographic position was not constant across all timber-harvest treatments or BGC zones, but instead involved combinations of these factors. Of the 14 "streamside" species, five were virtually limited to riparian sites, three were "edge" species that were significantly more abundant in streamside than upland clearcuts or corridors, four were forest species that were more common throughout ESSF forests but only common in the MS Zone in riparian forests (discussed in 3.4.4), and two were species where the situation was more complex. One species was not counted within the 16 species showing topographic position effects, but is actually a riparian-obligate species, the Northern Waterthrush. No significant association could be shown here because it appeared to respond to wet microhabitats within the 5-ha plots, rather than just to large-scale effects, and this resulted in a high coefficient of variation.

Of the two "upland" species, only Red Crossbills were significantly more abundant in upland forests than in riparian forests in general. Pine Siskins were significantly more abundant in upland control forests than in riparian control forests, but this situation did not hold in forest corridors.

The five species mostly limited to riparian sites within this investigation were Hammond's Flycatcher, Lincoln's Sparrow, Song Sparrow, Willow Flycatcher, and Winter Wren. Willow Flycatchers were limited to riparian sites but were generally very uncommon in streamside sites compared to their abundance in wetlands (see section 3.4.5).

The other four species were not necessarily limited to riparian sites over their entire range, but within the MS and ESSF zones studied here, these were their primary associations. Where some of these species were occasionally detected in upland habitat, they were found in each case to be using ephemeral streams, or small pothole swamps or marshes that were much smaller (i.e., <0.1 ha) than the 5-ha scale of the study plots.

The three "edge" species that were more abundant in riparian situations were the Brown-headed Cowbird, MacGillivray's Warbler, and Wilson's Warbler. MacGillivray's Warbler and Wilson's Warbler appeared to associate primarily with shrubs — and these were consistently present and more abundant at streamside than upland sites. Where tall shrubs (≥2 m) were abundant in upland clearcuts, these species were usually present. Brown-headed Cowbirds were simply more abundant in streamside clearcuts and corridors than upland clearcuts and corridors. In one case, a flock of Brown-headed Cowbirds was using the streamside habitat more than the upland because cattle were using this habitat more, and the cowbirds were perching on and foraging near the cattle.

For two abundant species, the situation was more complex. Dark-eyed Juncos were found in all habitats, but were significantly more abundant in riparian corridors than in upland corridors. For this ground-nesting species, the greater herbaceous ground cover found in riparian compared to upland forests (see Vaartnou 1994) might have explained this difference, except that this response by juncos did not hold within clearcuts as well, where they were as abundant in riparian as in upland clearcuts. Yellow-rumped Warblers were more abundant in upland control forests than riparian control forests, but this reversed within clearcuts where they were more abundant in riparian than upland situations. The higher abundance in streamside clearcuts compared to upland clearcuts might be explained by the greater amount of remnant understorey conifers in streamside clearcuts compared to the almost complete lack of these in upland clearcuts. Within forests, Yellow-rumped Warblers seemed to prefer more open situations, as in corridors or the lodgepole pine stands of the upland MS.[1]

3.4.4 Effects of biogeoclimatic zone

Of the 33 common species analyzed, 12 species showed significant effects due to associations with the MS-ESSF contrast studied here (Table 8). Only one species, the Boreal Chickadee, was found only in the ESSF. Four species were more abundant in the ESSF than in the MS: Evening Grosbeak, Hermit Thrush, Pine Siskin, and Varied Thrush. Only the Orange-crowned Warbler was more abundant in the MS than in the ESSF.

The Golden-crowned Kinglet, Red-breasted Nuthatch, and Townsend's Warbler were significantly less abundant in MS upland forests than in MS riparian forests and all ESSF forests. The trend was similar, but not significant, for the Varied Thrush. It was absent from MS upland forests, but so uncommon within MS riparian forests that no significant difference could be shown. The MS riparian forests appeared to be downward extensions of the spruce and subalpine fir forests typical of the ESSF, and these bird species appeared to respond to that habitat difference.

Within recently made forest corridors, Ruby-crowned Kinglets and Hermit Thrushes appeared to follow the same habitat preferences for spruce-subalpine fir forest over lodgepole pine forest as discussed for the previous four species. Ruby-crowned Kinglets were most abundant in corridors, rather than in continuous forest, and were significantly less abundant in MS upland lodgepole pine corridors than in MS riparian corridors or ESSF corridors. The results for Hermit Thrushes suggest the same trend since they were least abundant in MS upland forest and in all recent clearcuts. However, their abundance in other habitats, including regenerating clearcuts with >2-m tall conifers, statistically obscured this comparison.

The abundance of the Dark-eyed Junco was highest in clearcuts and corridors, except in ESSF corridors for which there was no clear explanation. The Yellow-rumped Warbler achieved higher abundance in MS corridors than in ESSF corridors, and this may be due to canopy differences in lodgepole pine and spruce forests (see section 3.4.3).

3.4.5 MS wetland riparian investigation

A total of 1733 birds was detected within the nine 5-ha plots in this investigation. Analyses were completed individually for 20 species of birds, while an additional 50 species were considered uncommon and could not be analyzed separately (see Appendix E). Overall bird abundance was higher in the wetlands than in the areas bordering the wetlands at every one of the nine plots. Total bird abundance was higher in the wetlands with some timber harvesting adjacent to them than in the control wetlands with no adjacent timber harvesting (Table 9).

Of the 20 species for which analyses were completed, 10 were significantly more abundant in the wetland than in the area bordering the wetland, and this was independent of any timber-harvesting effects in the bordering area (Table 9): Common Snipe, Common Yellowthroat, Lincoln's Sparrow, Mallard, Northern Waterthrush, Song Sparrow, Warbling Vireo, Willow Flycatcher, Wilson's Warbler, and Yellow Warbler. Only Common Snipe had higher densities over clearcuts than forests beside wetlands. This may be a real effect, or may be because its flight displays over clearcuts were detected more easily when the bird could be seen. When a flight display is over a forest, the exact location of the sound is not easily placed, and the observer could assume the display was over the wetland. One other species, Red-winged Blackbird, was effectively limited to wetlands, although the association was not statistically significant because of a high coefficient of variation. The high spread in the data resulted because it was not present in every wetland, but was extremely abundant where it was present. Timber harvesting in the area bordering the wetland also appeared to have no negative effect on Red-winged Blackbirds.

Five of the 20 species were not significantly different in abundance between the wetland and the bordering area (Table 9). No significant effects at all could be shown for the Orange-crowned Warbler, Pine Siskin, and Tree Swallow. However, the American Robin and Brown-headed Cowbird were equally abundant in the wetland and bordering area, but were significantly less abundant in the control forests adjacent to wetlands than when the borders were clearcut or had corridors of forest left along them.

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[1] Only canopy tree-stem densities were measured in this study (see section 2), and they were significantly lower in riparian than in upland stands. However, this may not hold for canopy cover or canopy volume, since the riparian trees were often more full-canopied because they tended to be grown in more open situations, and they were more often spruce than lodgepole pine.

Table 9. Significant factor effects for common bird species and total abundance of birds in the Montane Spruce wetland investigation. Full data and ANOVA statistics can be found in Appendix C.

COMMUNITY PARAMETERS	Significant differences between treatments (Mean no. detected in 2.5-ha plots)
Total abundance	Significant two-way interaction effect Control wetland (73.0) > Control border (34.0) Corridor wetland (162.7) > Corridor border (74.7) Clearcut wetland (178.7) > Clearcut border (54.7) Clearcut wetland (178.7) and Corridor wetland (162.7) > Control wetland (73.0)
SPECIES	
American Robin	Clearcut (3.8) > Control (0.3), but Corridor (2.3) not significantly different from either
Brown-headed Cowbird	Corridor (5.0) > Control (0.8), but Clearcut (3.2) not significantly different from either
Common Snipe	Wetland (2.6) > Border (0.6) Clearcut (3.2) > Control (0.5) and Corridor (1.0)
Common Yellowthroat	Wetland (16.0) > Border (0.4)
Dark-eyed Junco	Border (5.2) > Wetland (2.4)
Golden-crowned Kinglet	Absent from wetland Control (5.0) and Corridor (5.3) > Clearcut (0)
Lincoln's Sparrow	Wetland (10.2) > Border (1.8)
Mallard	Wetland (7.7) > Border (0.2)
Northern Waterthrush	Wetland (5.8) > Border (1.6)
Orange-crowned Warbler	No significant effects
Pine Siskin	No significant effects
Ruby-crowned Kinglet	Significant two-way interaction effect Corridor border (11.7) > Corridored wetland (0.7) Corridor border (11.7) > both Control border (2.3) and Clearcut border (0.3)
Red-winged Blackbird	No significant differences
Song Sparrow	Wetland (19.3) > Border (0.9)
Tree Swallow	No significant effects
Warbling Vireo	Wetland (3.6) > Border (0.4)
Willow Flycatcher	Wetland (13.6) > Border (0.4)
Wilson's Warbler	Wetland (7.9) > Border (1.8)
Yellow Warbler	Wetland (5.4) > Border (0.1)
Yellow-rumped Warbler	Significant two-way interaction effect Corridor Border (8.3) > Corridored Wetland (1.3) Corridor Border (8.3) > Clearcut Border (1.7), but Control Border (4.0) not significantly different from either

Four of the 20 species were significantly more abundant in the area bordering the wetland than in the wetland (Table 9). The abundance of the Dark-eyed Junco appeared largely unaffected by timber-harvest treatment in the border. Golden-crowned Kinglets were equally abundant in any forest bordering the wetlands, but absent in the wetlands or in the clearcut. Both Ruby-crowned Kinglets and Yellow-rumped Warblers were more abundant in the corridors of forest that bordered the wetland than in the wetland, in the clearcut, or in unharvested border situations.

3.4.6 Effects of corridor width

Six common forest species were found to be absent from forest corridors below certain widths, as listed in Table 10, and were termed "width-sensitive species." Density of the width-sensitive species group only declined compared to uncut forest as forest corridor width decreased below 72 m, and they were completely absent from corridors <38 m wide (Table 11).

Total density of bird detections was highest in the narrowest range (15-31 m) of MS riparian and ESSF corridors examined (Table 11), despite any decline in width-sensitive species. These total densities were highest in this range of forest corridors because of significantly greater densities of open-forest species, generalist, and other species than in uncut forest or wider corridors.

The narrowest range of riparian forest corridors studied here (15-31 m) only sometimes supported other common forest species and did not support any of the other uncommon forest species. At least some of the other common forest species were present in all of the corridors wider than 38 m. However, within this narrowest range (15-31 m), two of the four sites had none of these species, and the third site was represented in this group by only one detection of one Red Crossbill. The fourth site contained Varied Thrushes, Golden-crowned Kinglets, and Red-breasted Nuthatches, although these birds appeared to be moving back and forth to the more extensive floodplain forest only 50 m from the end of the sampled corridor. This disparity resulted in a coefficient of variation for this width class that was more than five times the mean for the other six width classes and 3.3 times the next largest coefficient of variation. It therefore appeared that riparian forest corridors narrower than 38 m did not have much value for bird species that are typically limited to closed canopy forest.

Table 10. Width-sensitive common forest bird species as determined by absence from narrow forest corridors.

Species	Width (m) of narrowest forest corridor in which the species was detected
Boreal Chickadee	103 m
Brown Creeper	50 m
Gray Jay	45 m
Swainson's Thrush	110 m in MS upland; Used all corridors in riparian
Townsend's Warbler	138 m in MS upland; 47 m in MS riparian and ESSF
Winter Wren	47 m

If edge effects within a corridor are defined as the point at which the densities of any species group significantly differed from unharvested forests (i.e., corridor width >500 m), then edge effects were found even in the widest corridors examined (class range 132-280 m, mean=188 m; see Table 11). Open-Forest birds (particularly the Yellow-rumped Warbler and Ruby-crowned Kinglet) were significantly more abundant in this corridor class than in control forests. Other species were found at significantly higher densities than in control forests in corridors <66 m wide. Using the proportion of mean total bird density of open-forest and other species as a relative measure of edge effect, then 21% is a baseline measure since these species made up 21% of the mean control forest bird density. These species made up 41-48% of the mean bird density in corridors 72-280 m wide, 58-60% of the mean bird density in corridors 38-66 m wide, and 69% of the density in corridors 15-31 m wide. Therefore, while edge effects do extend into the widest corridor class (132-280 m), there appeared to be a threshold level at about 70 m where the effect became very pronounced and "edge" or "clearcut" species made up more than 50% of the total bird density.

Upland MS forest corridors were examined separately from other forest corridors. Total density of bird detections was higher in narrow (39-50 m) MS upland forest corridors than in wider (110-188 m) forest corridors or in control forest (Table 11). Width-sensitive species (Brown Creeper, Swainson's Thrush, and Townsend's Warbler) were uncommon in MS upland forest to begin with, so that their absence from the narrowest range of corridors examined could not be shown to be significantly different from control forests or wider corridors.

Table 11. Density of total bird detections by species groups in forest corridors of different widths.

		No. detected/forested ha[1]							ANOVA statistics		
		Corridor width range (m)									
Species group	Width (N)	>500 (18)	132-280 (5)	104-125 (7)	72-86 (5)	50-66 (5)	38-49 (4)	15-31 (4)	$F_{6,41}$	P[2] (α)	Effect size[3] (f)
MS RIPARIAN AND ALL ESSF FOREST CORRIDORS											
Total density	Mean	2.92^{a}	4.16^{ab}	4.71^{ab}	5.57^{b}	5.79^{b}	4.47^{ab}	10.79^{c}	13.86	**0.000**	1.32
	S.E.	0.29	0.29	0.75	0.63	0.60	0.68	1.73			
Width-sensitive species[4]	Mean	0.80^{a}	0.87^{a}	0.87^{a}	0.78^{a}	0.48^{ab}	0.12^{bc}	0^{c}	4.82	**0.000**	0.78
	S.E.	0.11	0.02	0.13	0.21	0.12	0.12	-			
Other common forest species[5]	Mean	0.92	0.64	1.02	1.12	0.85	0.58	0.81	0.63	0.70	0.28
	S.E.	0.12	0.16	0.12	0.20	0.12	0.09	0.67			
Other uncommon forest species[6]	Mean	0.06	0.01	0.08	0.07	0.07	0.02	0	0.77	0.60	0.31
	S.E.	0.02	0.01	0.04	0.06	0.02	0.02	-			
Open-forest species[7]	Mean	0.54^{a}	1.63^{b}	1.49^{b}	2.18^{bc}	2.66^{c}	2.17^{bc}	5.20^{d}	26.84	**0.000**	1.83
	S.E.	0.06	0.18	0.34	0.23	0.30	0.37	0.87			
Generalist species[8]	Mean	0.54^{a}	0.63^{a}	0.82^{a}	0.95^{a}	0.87^{a}	1.16^{a}	2.01^{b}	5.45	**0.000**	0.83
	S.E.	0.07	0.11	0.21	0.14	0.24	0.40	0.47			
Other species[9]	Mean	0.07^{a}	0.36^{ab}	0.43^{ab}	0.48^{ab}	0.84^{b}	0.42^{ab}	2.22^{c}	8.77	**0.000**	1.05
	S.E.	0.02	0.15	0.19	0.11	0.18	0.11	0.87			

		No. detected/forested ha[1]			ANOVA statistics		
		Corridor width range (m)					
Species group	Width (N)	>500 (5)	110-188 (5)	39-50 (4)	$F_{6,41}$	P[2] (α)	Effect size[3] (f)
MS UPLAND FOREST CORRIDORS							
Total density	Mean	1.78^{a}	2.30^{a}	4.89^{b}	10.47	**0.003**	1.22
	S.E.	0.16	0.15	0.98			
Width-sensitive species[4]	Mean	0.17	0.20	0	2.55	0.12	0.60
	S.E.	0.05	0.09	-			
Other forest species[5, 6]	Mean	0.70	0.52	0.74	0.85	0.45	0.35
	S.E.	0.13	0.13	0.16			
Open-forest species[7]	Mean	0.41^{a}	0.84^{ab}	1.20^{b}	6.36	**0.015**	0.91
	S.E.	0.04	0.13	0.16			
Generalist species[8]	Mean	0.45^{a}	0.50^{a}	1.85^{b}	6.81	**0.012**	0.99
	S.E.	0.04	0.28	0.57			
Other species[9]	Mean	0.05^{a}	0.24^{a}	1.10^{b}	18.97	**0.000**	1.65
	S.E.	0.02	0.13	0.19			

[1] Where ANOVA significant, means followed by like superscripts not significantly different using Duncan's Multiple Range Test.

[2] Significant α denoted in **bold**.

[3] Effect Size (f) of Cohen (1988), where 0.4 denotes large effect, 0.25 medium effect, and 0.1 small effect.

[4] Width-sensitive forest species: Boreal Chickadee, Brown Creeper, Gray Jay (in riparian), Swainson's Thrush (in upland), Townsend's Warbler, Winter Wren.

[5] Other common forest species: Evening Grosbeak, Golden-crowned Kinglet, Gray Jay (in upland), Red-breasted Nuthatch, Red Crossbill, Varied Thrush.

[6] Other uncommon forest species: Black-backed Woodpecker, Hairy Woodpecker, Nashville Warbler, Pine Grosbeak, Solitary Vireo, Spruce Grouse, Three-toed Woodpecker, White-breasted Nuthatch, Western Tanager, Western Wood-Pewee.

[7] Open-forest species: Cassin's Finch, Fox Sparrow, Hammond's Flycatcher, Hermit Thrush, Mountain Chickadee, Northern Waterthrush, Olive-sided Flycatcher, Red-naped Sapsucker, Red-tailed Hawk, Ruby-crowned Kinglet, Swainson's Thrush (in riparian), Wilson's Warbler, Yellow-rumped Warbler.

[8] Generalist species: Dark-eyed Junco, Pine Siskin.

[9] See Appendix G for list of other species.

Table 12. Density of bird detections by species groups in partially harvested streamside forest corridors compared to totally reserved streamside forest corridors.

	No. detected/forested ha (N=4 for each mean)				ANOVA statistics			
	Total reserve		Partial harvest				Effect	
Species group	Mean	S.E.	Mean	S.E.	$F_{1,6}$	P[1] (α)	size[2] (f)	Power (1-β)
WIDE RIPARIAN FOREST CORRIDORS (69-150 M) AND LOW LEVELS OF TIMBER HARVEST (16-32%)								
Total density	5.40	0.30	6.55	1.07	1.07	0.34	0.37	0.14
Width-sensitive species[3]	1.06	0.16	0.58	0.18	3.89	0.10	0.70	0.38
Other forest species[4]	0.97	0.17	1.40	0.30	1.56	0.26	0.44	0.18
Open-forest species[5]	1.60	0.10	2.00	0.31	1.46	0.27	0.43	0.18
Generalist Species[6]	1.08	0.21	1.50	0.58	0.44	0.53	0.23	0.09
Other Species	0.69	0.07	1.09	0.33	1.41	0.28	0.42	0.17
NARROW RIPARIAN FOREST CORRIDORS (47-55 M) AND HIGH LEVELS OF TIMBER HARVEST (48-85%)								
Total density	5.24	0.97	7.73	1.39	2.15	0.19	0.52	0.24
Width-sensitive species	0.39	0.13	0	-	8.85	**0.025**	1.05	0.70
Other forest species	0.64	0.18	0.25	0.12	7.47	**0.034**	0.97	0.63
Open-forest species	2.31	0.48	3.07	0.51	0.76	0.42	.31	0.12
Generalist species	1.04	0.10	1.53	0.51	0.89	0.38	.33	0.12
Other species	0.65	0.25	2.88	0.60	11.67	**0.014**	1.21	0.81

[1] Significant α denoted in **bold**.

[2] Effect size (f) of Cohen (1988), where 0.4 denotes large effect, 0.25 medium effect, and 0.1 small effect.

[3] Width-sensitive forest species: Brown Creeper, Gray Jay, Townsend's Warbler, and Winter Wren.

[4] Other forest species: Evening Grosbeak, Golden-crowned Kinglet, Hairy Woodpecker, Nashville Warbler, Red-breasted Nuthatch, Red-eyed Vireo, Red Crossbill, Solitary Vireo, Spruce Grouse, Three-toed Woodpecker, Varied Thrush, White-breasted Nuthatch, and Western Wood-Pewee.

[5] Open-forest species: Cassin's Finch, Fox Sparrow, Hammond's Flycatcher, Hermit Thrush, Mountain Chickadee, Northern Waterthrush, Olive-sided Flycatcher, Red-naped Sapsucker, Red-tailed Hawk, Ruby-crowned Kinglet, Swainson's Thrush, Wilson's Warbler, and Yellow-rumped Warbler.

[6] Generalist species: Dark-eyed Junco and Pine Siskin.

3.4.7 Effects of partial timber harvesting

Open-forest species and other species made up 42% of the mean total bird density in uncut wide (69-150 m) streamside forest corridors, and 47% in corridors of the same width range but with low levels (range of 12-32% removal by tree-stem count, mean=24%) of tree-stem removal (Table 12). The levels of forest bird species (width-sensitive and other forest species) were not significantly lower when partial timber harvesting was only at these low levels. The effects of partial timber harvesting on bird densities at these low levels of removal appeared to be minimal.

The effects of high levels of partial harvest (range of 48-85% removal by tree stem count, mean=69%) in narrow streamside forest corridors (47-55 m) were in marked contrast to the minimal effects of low levels of partial harvest in wider corridors (Table 12). Width-sensitive species were completely absent from the corridors with high levels of partial harvest, and other forest species were significantly lower in density. Other species were significantly higher in density in the corridors with high levels of partial harvest. Open-forest species and other species made up 56% of the mean total bird density in these uncut corridors of 47-55 m, but 77% of the total density in corridors of the same width, but with a high level of partial removal. In terms of creating an edge effect (as measured by the proportion of total density made up of other species and open-forest species), the effects of high levels of partial removal in 47-55-m wide corridors is even stronger at 77% than the effects of leaving a very narrow corridor reserve of 15-31 m, where these species make up 69% of the total density (see Table 11).

With only two replicates (50% and 11% removal) of partially harvested MS upland forest corridors, no separate analysis was possible. However, there appeared to be very little effect of partial cutting in these narrow corridors, so these plots were included within the of MS upland corridor width analysis.

3.5 Discussion

3.5.1 Classification of bird species by habitat and response to timber harvest

The results of the first two investigations, additional analyses on corridor widths and partial harvesting, and casual observations in the study area allowed classification of most of the bird species observed according to their habitat preferences. The first two investigations identified 91 species of birds (see Appendices D and E). An additional three species were identified on additional corridor width and partial harvesting plots (Common Nighthawk, Black-backed Woodpecker, and White-breasted Nuthatch); two additional species on internal validation bird-sampling transects (Common Loon and Rock Wren, see section 7) and three species were identified within the May-July breeding period, but not on any sampling plots (Cooper's Hawk, Golden Eagle, and White-winged Crossbill). White-winged Crossbills depend on conifer cone-crops and do not necessarily breed during the flush of insect abundance in the summer. They were never detected in the area during May and June, but were commonly detected singing in MS riparian spruce forests in the study area in July 1995 after the breeding-bird sampling was finished. The 101 species listed in Appendix F probably form a relatively complete list of the breeding songbirds in the MS and ESSF BGC zones of the study area. Coverage was certainly less complete for owls and probably for diurnal raptors, shorebirds, and waterfowl as well.

Of the 101 species, 37 (37% of total species, see Appendix F) were associated strongly with riparian areas compared to only one species (Red Crossbill) that was more abundant in upland forests. Twenty-two of these riparian species were more dependent on open water, marshes, or shrub swamps than on forests and would not be directly affected by timber harvest. Nine riparian species would be affected by timber harvest because they depend on trees for nesting or as foraging perches along streams or wetlands, or depend on healthy fish or stream invertebrate populations (which can be affected by poor logging practices). The remaining six species were dependent on riparian forests within the MS Zone because these were the only stands in the MS where spruce predominated. Only the Winter Wren was also dependent on riparian forest in the ESSF Zone.

Of the 101 species, seven (7% of total) were swifts and swallows, which appeared to spend a considerable amount of time foraging in wetlands, but have not been specifically identified as riparian species here. These species are highly mobile and often forage considerable distances away from the nests (up to 5.4 km in the case of Vaux's Swift; Bull and Beckwith 1993). Habitat associations can often be obscured, rather than clarified, if based on 5-ha plots, which are small compared to the breeding territories of these birds.

Including the swifts and swallows, 64 species (63% of total) did not depend on riparian areas, even though all these species appeared to make use of either wetlands or riparian forests as well as upland clearings or forests. Twenty-six of these other species had little or no requirement for forest habitats, 20 would not be as strongly affected by timber harvest, since they did not appear to require closed canopy forest, but did require some forest elements, and 18 appeared to require closed forest habitat. These swifts, swallows, and other species were not considered further here, since their management potential was not linked specifically to forest practices in riparian areas. However, even though they will not be further considered, management of riparian forests could contribute significantly to the habitat needs of these species in the absence of suitable forest in adjacent upland areas.

3.5.2 Wetland species

The most common species of wetland birds did not appear to be affected by timber harvest adjacent to the wetlands. If anything, the opposite was true — many typical wetland birds seemed to prefer open habitat adjacent to the wetland rather than riparian forest, and only used streamside habitats if clearcutting or heavy partial cutting had been to the edge of the stream bank. Croonquist and Brooks (1993) also found wetland-dependent birds were more abundant in disturbed than undisturbed sites. Many of the most common wetland species found here, including Song Sparrow, Yellow Warbler, Lincoln's Sparrow, and Willow Flycatcher, are typical of willow thickets and marshes in the completely treeless sagebrush steppes of western North America (Knopf et al. 1988b), and they will probably not be negatively affected by timber harvest adjacent to wetlands. The same may be true of the other 18 species identified as riparian-dependent that were unlikely to be negatively affected by timber harvest (see Appendix F). In this study, these birds apparently did not rely on forest adjacent to the wetlands.

3.5.3 Riparian species needing trees

Eight riparian-dependent species that would be affected by timber harvest because they needed suitable nesting or perching trees were identified. Barrow's Goldeneye, Bufflehead, and Black-capped Chickadee are cavity nesters, and American Crows build open nests in trees. Black-capped Chickadees and American Crows are not necessarily limited to riparian areas, but in the study area, these are the only places they were found. Belted Kingfishers require fish to feed on, as well as perch trees with a recommended spacing of at least one per 25 m (Prose 1985).

Great Blue Herons and Ospreys nest in trees (alive or dead) that have limbs big enough to support their large nests. These special nesting trees are not necessarily common, and are not necessarily found within a narrow band adjacent to a lake or wetland. Only one Osprey nest was found within this study, and it was in a coniferous swamp that was sampled for amphibians. The nearest feeding site for the Osprey was Grizzly Lake, which was 500 m distant.

Northern Waterthrushes are a riparian-obligate species that require standing water, but do not necessarily require standing trees of full canopy height. They were often found in stagnant old stream channels or pothole swamps at least 0.1 ha in size where there were high levels of shrubs, usually some live conifers, and often standing dead trees. Northern Waterthrushes were 3.3 times more abundant on the riparian forest plot that had been windthrown, and then logged by helicopter, than on any other plot. Clearcut riparian forests retained habitat suitability for this species only if many tall shrubs and understorey conifers, approaching 50% total cover or greater, were retained on the site.

3.5.4 Riparian species feeding on stream invertebrates

Only one species in the study area specializes in feeding on stream invertebrates. American Dippers were very uncommon in the study area and were only detected four times on three streams (Belgo Creek, Trapping Creek, and Copperkettle Creek) within the 21-km total stream length sampled within this study. Most streams on the plateau appeared to be too narrow (<2 m wide) to support dipper populations (Tyler and Ormerod 1994). No dippers were encountered during internal validation sampling of all the streams in two small drainages (total length 7.9 km; see section 7).

Dippers, however, do not appear to require standing streamside forests. American Dippers can be numerous on subalpine streams that do not necessarily have continuous tree cover adjacent to the streams, and even breed within the city limits of Kelowna along Mission Creek, where the stream channel is diked and the degree of tree cover adjacent to the dikes is variable (L. Gyug, pers. obs.). Effects of timber harvesting on dippers are likely to be indirect and related to healthy stream insect populations, rather than directly tied to canopy cover adjacent to streams.

3.5.5 Riparian species requiring forests

The fourth major group of riparian-dependent species were those that required spruce-subalpine fir forests. In the MS Zone, spruce forests predominated adjacent to streams and wetlands, although spruce forests were prevalent throughout the ESSF. Even where lodgepole pine dominated in the upland ESSF, there were always more spruce and subalpine fir within those stands than in the upland MS (see section 2). MS streamside stands are habitat extensions for birds generally more typical of the ESSF than the MS. For five species identified here (Golden-crowned Kinglet, Hammond's Flycatcher, Townsend's Warbler, Varied Thrush, and White-winged Crossbill), this means that in the MS they will only be abundant (or found at all) in forests adjacent to streams and wetlands, although they may be widespread and abundant in the ESSF. Only the Winter Wren was fairly limited to riparian forests in both the MS and ESSF.

For the six species identified as being dependent on riparian forests in the MS, the habitat differences may be explained by a preference for spruce over pine. Franzreb (1984) found this habitat correlation for Golden-crowned Kinglets. The preference, however, may not strictly be for spruce, but for coniferous trees with larger canopy volumes or denser canopies than lodgepole pine, or for forests with more complex understories than lodgepole pine forests. Each of these species (including the Winter Wren) can also be abundant in upland west-coast forests where spruce may be absent or not abundant (e.g., Wetmore et al. 1985; McGarigal and McComb 1992), so that the habitat feature(s) required cannot be simply the presence of spruce.

3.5.6 Biogeoclimatic zones

In general, the MS lodgepole pine forests appeared to have a less diverse bird community than ESSF spruce-subalpine fir forests — principally because of the species that will rarely use the MS upland lodgepole pine forests. A similar trend has been noted in other areas of the western cordillera dominated by lodgepole pine forests, such as Banff and Jasper national parks in Alberta, where Holroyd and Van Tighem (1983) found that Varied Thrushes, Golden-crowned Kinglets, and Townsend's Warblers all reached their highest densities in closed spruce-fir forest.

It is not certain how the effects of riparian forests in providing habitat extensions for typical ESSF birds may continue down into biogeoclimatic zones below the MS (particularly the Interior Douglas-fir and Ponderosa Pine zones). However, since spruce occurs commonly in the riparian forests of these lower elevations and is absent from upland forests, this effect may continue downward through the entire elevational gradient.

3.5.7 Effects of corridors

Riparian corridors — In this study, Townsend's Warbler, Winter Wren, Brown Creeper, Boreal Chickadee, and Gray Jay were found to be sensitive to forest-corridor width, with significant declines in density in corridors narrower than 72 m. When the forest-corridor width fell below 50 m, the habitat suitability disappeared entirely for these species. Suitability for all other forest bird species requiring closed-canopy forests disappeared when forest-corridor width fell below 38 m. For streams, forest corridor width included both banks, so that if interpreted in terms of "stream buffer-zone width," the figures could be halved. However, next to wetlands, these are buffer-zone widths.

The riparian forest corridors that supported forest species did not need to be all floodplain or valley bottom to provide suitable habitat. As long as the total forest widths were achieved, the floodplains supporting spruce forest could be as narrow as a few metres. In these cases, the narrow linear patches of spruce were surrounded by forest that was more similar to upland lodgepole pine forest than to riparian forest. Retention of a very narrow (anything <38 m) spruce floodplain forest alone was not sufficient to provide habitat for these species, even if the surrounding forest prior to clearcutting had been upland lodgepole pine. Retention of forest-corridor width, with some spruce at its core, appeared to be required to maintain habitat suitability. However, whether this habitat suitability comes from a requirement for certain microclimatic conditions in the riparian forests or from requirements to forage into larger forested areas from the (sometimes) narrow streamside core habitat is not known.

Besides the importance of riparian forests in the MS to typically ESSF birds, these riparian forests became important to a much wider variety of species when the areas around them are clearcut. At that time, the riparian forest corridors become as important to the avifauna in general as the gallery riparian forest corridors of the dry southwestern U.S. (e.g., Knopf and Samson 1994). In both cases, the forests along the streams are linear habitat patches surrounded by habitat that is generally lacking in vertical structure. This importance in the Okanagan Highlands was reflected in the highest abundance of birds being found in the riparian forest corridors. The importance of these riparian forest corridors to a variety of species cannot be replaced by upland forest corridors if the upland corridor is primarily lodgepole pine (see "Upland corridors" below).

Some eastern North American riparian forest birds have been shown to be highly sensitive to riparian forest-corridor width. Croonquist and Brooks (1993) found sensitive species to be those that required at least 25 m of undisturbed forest on each bank of the stream for a total corridor width of 50 m. Unfortunately, they did not list these species, so specific results cannot be compared between studies. Keller et al. (1993) found several species for which the probability of occurrence in riparian forests increased with increasing total corridor width. Of these species, only the Red-eyed Vireo occurs in the Okanagan Highlands, but was too uncommon to determine any corridor-width effect here. Stauffer and Best (1980) found many species sensitive to riparian forest-corridor width, five of which also occur in the Okanagan Highlands. The Black-capped Chickadee (15 m reported minimum width), White-breasted Nuthatch (17 m reported minimum width), Hairy Woodpecker (40 m reported minimum width), and Red-eyed Vireo (40 m reported minimum width) were too uncommon in the study area to determine any effects. Warbling Vireo (90 m reported minimum width) was found in MS clearcuts where patches of deciduous trees or tall shrubs had been retained, so that dependence on any forested corridor was not evident here.

Upland corridors — For many species, the habitat suitability of MS lodgepole pine upland forests differed markedly from MS riparian and all ESSF forests. Extensive corridor-effect analyses were not made for upland lodgepole pine forests as were made for the riparian forests, because not as many upland corridors occurred in the study area. With the high potential of narrow upland corridors for windthrow and as

Mountain Pine Beetle infestation centres, forest licensees did not create these sites when timber harvesting. Typical reserves in upland clearcuts tended to be small tree patches on rock outcrops rather than narrow forest corridors with arbitrary boundaries.

Within the narrowest range of MS upland corridors sampled (39-50 m), no forest species needing closed canopy was reliably present on every visit — indicating that few or none of them may have been breeding there. The most common open-forest bird, and the only one reliably present at each plot, was the Yellow-rumped Warbler. These narrow upland corridors were extremely silent compared to the riparian forest corridors of similar width. Three of the corridors were within hearing distance (<400 m) of riparian forest corridors. When sampling the upland corridors in calm conditions, it was possible to count more birds and more species singing from the riparian corridors than from the upland corridor in which the observer was standing. This is one case where using all detections, rather than just singing detections, may have biased results, so that total bird densities may have looked fairly comparable to similar-width riparian corridors (Table 11). One-third of the observations in the upland corridors were of Dark-eyed Juncos, which only made up 3-11% of the observations in forests or forest corridors, and only 8-25% of the observations in clearcuts. These upland corridors were clearly less diverse in species than any other type of site, and were dominated numerically by one of the most abundant (and least endangered) species in the study area.

Swainson's Thrushes were absent in upland forest corridors of <100-m width, even though they were abundant in heavily partially harvested riparian forest corridors of even very narrow widths. Swainson's Thrushes were probably relying on understorey tall shrubs and conifers; this understorey shrub and conifer layer is generally very sparse in the MS upland lodgepole pine forests. The lack of understorey may also explain the general paucity of species in the upland pine corridors in general.

3.5.8 Effects of partial timber harvest

Low levels of timber harvesting (<33% stem removal) appeared to have little effect on forest birds in riparian forest corridors, and would probably be the most efficient method of harvesting in riparian corridors to conserve the diversity of birds. Higher levels of harvesting (>48% stem removal) coupled with the effects of narrowness (<50-m width) on riparian forest corridors eliminated forest birds from the sites.

The effects of windthrow are sometimes comparable to the effects of partial timber harvesting. The net result of high levels of windthrow and high levels of partial harvesting in narrow corridors to bird habitat appeared to be similar: bird species requiring closed canopy forests were eliminated. The results presented in Table 12 for high removal partial harvest levels already incorporated the windthrow site with the partially harvested sites. However, not enough replicates were available to detect changes in abundance of single species.

Although windthrow and partial timber harvesting may create similar habitat for birds, windthrow did not imitate clearcut logging, even when >90% of trees were downed. In the one streamside plot that had been windthrown and then salvage-logged by helicopter, species composition changed after the salvage logging. Reconnaissance sampling had been done at the site in June 1993 (windthrow had occurred in October 1991), one sample was made in 1994 after the few remaining standing trees had been felled in preparation for salvage, and the site was fully sampled in 1995. When the few remaining standing trees were felled, several species disappeared from the site. In particular, Golden-crowned Kinglets, Hairy Woodpeckers, Hammond's Flycatchers, and Red-breasted Nuthatches used the site after windthrow, but not after salvage logging. Ruby-crowned Kinglets had been abundant and widespread after windthrow, but were only found in one small part of the site where a few trees had been left standing after salvage logging. After this, the plot was dominated by three species that appeared to thrive in the dense undergrowth: Northern Waterthrush (22% of detections), Wilson's Warbler (18%), and Dark-eyed Junco (15%). This particular combination of dominant species was found on no other plot (including wetlands) in the study.

Winter Wrens were sometimes detected on the site that was windthrown and then logged by helicopter, even though they were absent from narrow riparian forest corridors with heavy levels of partial cutting or windthrow. This could be due to the amount of understorey and woody debris retained on the helicopter-logged site. This site had a very wide floodplain (>200 m), and large amounts of downed woody debris, tall shrubs, and short conifers were present, even after salvage logging. Winter Wrens did make some use of high-lead logged clearcuts in west-coast hemlock forests (Wetmore et al. 1985), and even nested in a slash pile within a clearcut in Montana, where much understorey and many trees were left during timber harvest (Tobalske et al. 1991). If the helicopter-logged site in this study had been clearcut by conventional methods, and the timber groundskidded to landings, then fewer shrubs and small conifers would have remained, and the site would probably have had no suitability for Winter Wrens.

3.5.9 Power of the experimental design

Power of the experimental design to detect large effect sizes (f≥0.4; see Cohen 1988) were adequate (i.e., over 50%) for species in the MS-ESSF streamside investigation when there were no significant interactions of factors. This was because effective sample size was much larger when marginal means could be compared rather than cell means. Power of the MS wetland investigation was considerably lower with adequate power (>50%) only when effect sizes were ≥0.55 because there were only three replicates per cell. Power analyses were not done for the corridor-width comparisons because of differences in number of replicates between width classes, but power appeared to be even poorer yet, since only effect sizes >0.9 were significant at P(a) = 0.05. The implications of these power analyses for the experimental design are: (1) to detect large effect sizes for species, at least five replicates should be used for any treatment within a two-way or higher ANOVA (rather than just three as in the MS wetland investigation), and (2) in a one-factor ANOVA on corridor width or partial harvest level effects, >5 replicates per treatment are needed. If a similar study were to be repeated, or a study designed with timber harvesting done to experimental specifications, then these number of replicates could be used as initial targets.

3.6 Implications for Timber-Harvest Management

Managing solely for high densities or diversity at the stand level is not advocated for riparian forests because this "could have severe consequences for unique elements of riparian avifaunas" (Knopf and Samson 1994). Managing for stand-level effects may achieve local benefits while negating use of the habitat by certain species that require certain attributes not met in these stands. In the MSdm1 of the study area, managing for highest total bird abundance would imply creation of forest corridors of ±50 m total width along streams, perhaps with high levels of partial cutting, or even of helicopter clearcutting of floodplain forest. However, managing every riparian corridor for highest diversity would clearly negate their use by bird species that require forest.

Wetland species — On the other hand, bounding every stream with a wide forest corridor would negate the streamsides as habitat for riparian birds that do not use forest. However, most of these riparian-dependent birds achieve highest densities in natural wetlands, and in general, these species need not be considered in forest management because they are essentially non-forest birds. Under no present forest-management schemes will these species ever be short of habitat, particularly because many of these species can also be common along seepage sites and ephemeral drainages within clearcuts. However, these small sites will not be covered under the FPC riparian guidelines. These seepages and ephemeral drainages within clearcuts should receive special attention, i.e., understorey tree and shrub retention, to provide some habitat for these species, even if these areas do not fall specifically within the FPC riparian guidelines.

Riparian species needing trees — Barrow's Goldeneye, Bufflehead, Black-capped Chickadee, American Crows, and Belted Kingfishers should have their tree requirements met by FPC guidelines wherever reserves exist of at least 1.5 tree lengths along water bodies or wetlands, so that wildlife trees do not need to be felled to meet WCB regulations. For sites without reserve zones, wildlife tree maintenance will depend on "patchy" cuts within the Riparian Management Zone so that some wildlife trees can be retained for nesting. For Great Blue Herons and Ospreys, whose nests are usually large and conspicuous, buffer zones should be established separately around known nests or nests found during pre-harvest surveys, even if the nests are beyond the normally recommended reserve distance from a lake or wetland.

Riparian species requiring forests — Riparian forests are only critical habitat for a few species. Six species were identified that would be greatly affected by timber harvesting of riparian forests: the Winter Wren and Hammond's Flycatcher were fairly limited to riparian forests in the Okanagan Highlands; Golden-crowned Kinglet, Townsend's Warbler, Varied Thrush, and White-winged Crossbill[2] were only limited to riparian forest in the MS, since they occurred in ESSF forests of all types. If the FPC riparian guidelines were to reserve a minimum of 75 m total forest-corridor width along all streams, then the needs of all these species would be met. However, the guidelines only require reserves from 20 to 60 m wide.

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[2] The identification of the White-winged Crossbill is based on incidental observations, since they established territories only in July after sampling was finished. The data may not therefore be reliable enough to draw up any management scheme for this species.

Requirements are for 30-m reserves on each side of an S2 stream (5-20 m wide), 20 m on each side of an S3 stream (1.5-5 m wide), and the Best Management Practice is for 10 m reserve on each side of S4 streams (<1.5 m wide). S2, S3, and S4 streams are fish-bearing or within community watersheds. (There are no S1 streams in the study area.) No reserves are required on S5 and S6 streams on non-fish-bearing streams or streams outside community watersheds. The widest forest corridor under these guidelines will be 60 m, which may usually support these species, but this will only be applied to a minority of the streams in the study area. The smaller reserve zones will not be wide enough to support all these species reliably.

The Riparian Management Area (RMA) adjacent to the reserve zone cannot be counted on as habitat for these species unless that part of the area is considered active floodplain and retention is set at 70%. Given the narrow definition of active floodplain used in the FPC riparian guidelines (which is closer to "annual" floodplain than the geomorphologic definition of active floodplain), very few wide corridors will be created. Where the RMA is widened to "windfirm boundaries" to protect the reserve zone, the corridors may indeed become wide enough to support these species, but it is unknown how often this will be applied. Windthrow hazard can be considered high in almost all the riparian forests of the Okanagan Highlands because the spruce are typically shallow rooted on soils with high water tables. Wherever possible, the RMA boundaries should be widened to protect the windfirmness of floodplain forests to protect habitats needed by these forest bird species.

In the ESSF, management of habitat for these species (except the Winter Wren and Hammond's Flycatcher) will only need to consider forest-corridor widths, whether that corridor be upland or riparian, and variables such as patch sizes and overall areas of forest remaining. As such, the problems and solutions in the ESSF are not unique to riparian zones and will not be considered further here. Both Golden-crowned Kinglets and Varied Thrushes were found in very narrow (<38 m) riparian forest corridors, so that width effects will not be as critical for them as for width-sensitive species. The particular species that will require appropriate mitigation schemes for the effects of riparian forest harvest will be the Winter Wren and Hammond's Flycatcher, which are limited to riparian forests, and the Townsend's Warbler, which is fairly limited to spruce forests in the MS.

For Townsend's Warblers, a possible mitigative method would be to "short-circuit" the successional cycles, where lodgepole pine is slowly replaced by spruce over a 200 to 300-year period: spruce could be grown alongside pine in managed upland stands. Townsend's Warblers were abundant in some MS upland forests, but only in old-growth (>250 years old) forests or younger forests, where some spruce became established alongside the lodgepole pine that sprouted after the last stand-replacing fire. Such forests were not common within the MS, but did occur. Recommended target levels of spruce within such an MS upland forest cannot be precisely set at this time, but are probably in the range of 20 to 50%.

Mitigative measures should also be possible for Winter Wrens. It is not yet known at what stage of forest regeneration they will "return" to a site. Certainly, they will return to 80 to 100-year-old forests along streams as observed in this study, and can be abundant in 30 to 40-year-old "high-graded" riparian forests in the southern interior of B.C. (L. Gyug, pers. obs.). In coastal forests, they will sometimes occupy dense 30 to 50-year-old plantations at numbers similar to older forests (Hansen et al. 1995), although in other studies (McGarigal and McComb 1995; Bryant et al. 1993) they achieved only about half the old-growth densities in 30 to 60-year-old stands. In the Okanagan Highlands, Winter Wrens will probably be absent from managed riparian forests until a dense canopy forms after perhaps 40 to 50 years, and even then, only if stands retain a multi-layered effect with significant amounts of small conifers and shrubs and downed woody debris.

The FPC riparian guidelines have been constructed to minimize the effects of windthrow on riparian areas by removing windthrow-prone trees in preference to keeping them on S4, S5, and S6 streams. Typical nesting sites for Winter Wrens are in the large upturned rootballs of large trees (Harrison 1979) and in the tangles of downed woody debris typical of riparian forests, where trees easily fall over due to shallow rooting (observed in this study). It is unknown how effective the riparian management guidelines will be at controlling the effects of windthrow in riparian forests beside S4, S5, and S6 streams where windthrow hazard control is paramount. If the control is too efficient, then riparian areas may end up without some of the structural diversity necessary for Winter Wrens, and habitat suitability may be significantly reduced.

Hammond's Flycatchers did not appear to be as sensitive to either partial timber harvesting or riparian forest-corridor width as the other species considered here. In particular, they were more abundant along streams when the streams had been clearcut only on one side. However, in each case where they were detected in heavily partially harvested streamside forests, the individuals appeared to be maintaining territories that included more extensively forested streamside sites just beyond the sampled site. The riparian management guidelines are likely to meet the needs of this species along S2 and S3 streams. Along S4, S5, and S6 streams, habitat will only be maintained where the guidelines are not applied evenly over long stream reaches. For instance, if the S4, S5, or S6 streams were subdivided into 400- to 500-m reaches, and wider forest corridors were retained along every second reach with the regular guidelines applied to the other reaches, then habitat suitability might be maintained along most of the stream reach.

In summary, the FPC riparian guidelines will not maintain habitat suitability for a group of birds that rely on riparian forests in the Montane Spruce Zone of the Okanagan Highlands. However, the ultimate effects of the guidelines on these species will depend on when, or if, these birds will reoccupy clearcut riparian forests or narrow forest reserves as time passes. The effects will also depend on how the guidelines are applied over entire landscapes so that population bottlenecks do not occur that might limit emigration to habitat as it again becomes suitable. These effects remain to be investigated.

4 SMALL MAMMALS

4.1 Introduction

Very few studies have compared riparian small mammal communities to upland communities. Geier and Best (1980) documented the importance of streamside habitat to small mammals in the mid-west. Doyle (1990), McComb et al. (1993a, 1993b) studied riparian and upland small mammal communities in western Oregon. Small mammal abundance and species richness were either higher in riparian forests than upland forests, or not significantly different, depending on how different the forests were. Species composition also seemed to depend on the degree of differences in vegetation. No published studies in montane forest have specifically compared upland and riparian small mammal communities.

The specific objectives of this small mammal study were to:

- Determine small mammal species richness, composition, and abundance in upland and riparian areas under two timber-harvest regimes (clearcutting and reserve of forest corridors) compared to unharvested forest.
- Determine the effects of forest-corridor width and level of partial timber harvest in forest corridors on small mammal species richness, composition, and abundance.
- Make recommendations for timber-harvesting guidelines that would retain small mammal diversity in riparian areas.

4.2 Study Area

The study area was in the Montane Spruce (MSdm1) biogeoclimatic zone (Lloyd et al. 1990) of the Okanagan Highlands east of Kelowna, B.C. (Figure 9). The northern part of the study area was in the Vernon Forest District and the southern part was in the Penticton Forest District, both in the Okanagan Timber Supply Area of the Kamloops Forest Region of the B.C. Ministry of Forests.

The study area was chosen first because there were extensive existing clearcuts due to Mountain Pine Beetle salvage logging of older lodgepole pine forests that dominate the area, and second, the area was a plateau (1200-1400 m) with numerous riparian sites, including many low-gradient streams. The combination of these factors had produced a mosaic of timber-harvesting conditions adjacent to streams that had produced sufficient replicates to conduct this study. No sites were logged specifically for this study.

The forest vegetation of the area was dominated by a fire-caused disclimax of lodgepole pine in upland stands, which generally proceeds to a spruce (white-Engelmann hybrid) and subalpine fir climax in the absence of major fires. Spruce and subalpine fir were common at all stages of regeneration along streams and in wetter sites.

4.3 Methods

4.3.1 Study design

Comparison of clearcuts and corridors to controls — The experimental design was a two-factor analysis of variance (ANOVA). The first factor was topographic position with two levels selected: riparian streamside and upland. The second factor was timber-harvest treatment with three levels selected: controls where no timber harvesting had taken place, corridors where linear reserves of forest had been retained, and clearcuts where timber harvesting had removed all merchantable trees. Each unique combination of the

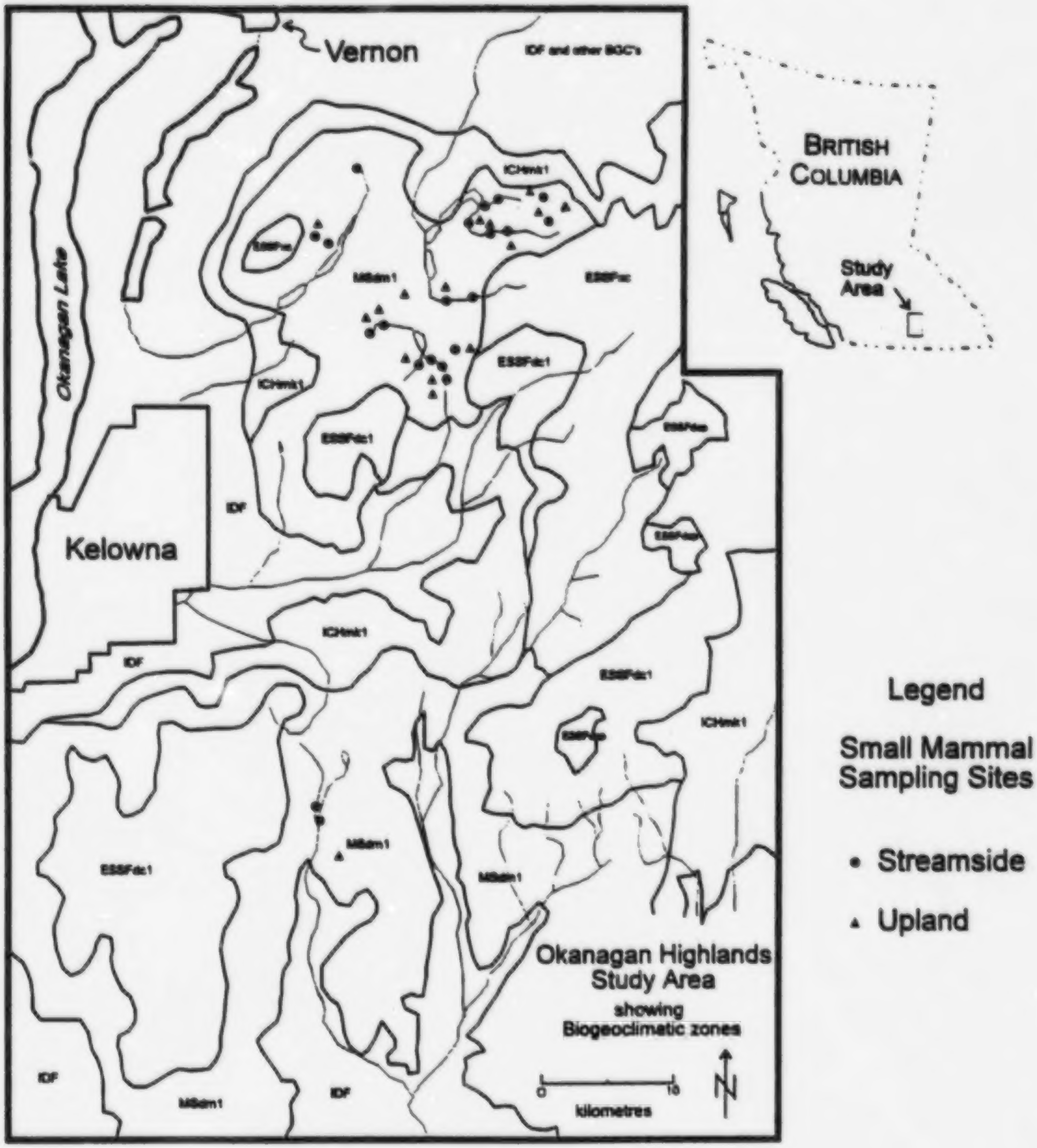

Vernon
IDF and other BGC's
British Columbia
Study Area
Okanagan Lake
ESSFwc
ICHmk1
MSdm1
ESSFwc
ESSFdc1
ICHmk1
ESSFdc1
ESSFwc
IDF
ESSFwc
Kelowna
ICHmk1
IDF
ESSFdc1
ESSFwc
ICHmk1
MSdm1
ESSFdc1
MSdm1
Okanagan Highlands Study Area
showing Biogeoclimatic zones
0
10
kilometres
N
Legend
Small Mammal Sampling Sites
• Streamside
▲ Upland

Figure 9. Small mammal study sites in the Okanagan Highlands.

factors was termed a treatment, with five replicates selected per treatment.

Sampling sites were selected from 1:15 000 air photos of the study area. Stream reaches were selected with a canopy gap of <10 m over the stream. Streams were S1 to S3. For unharvested and for clearcut sites, transects were sampled in pairs: one along the stream and one parallel to the stream between 100 and 200 m upslope, and always above the first major terrace. Corridors of forest reserved along streams and upland corridors of forest could not be sampled in the same way because upland corridors did not always exist parallel to riparian forest corridors. Forested corridor widths of sampled streamside replicates were between 62 m and 500 m, and 36 to 190 m of upland corridors. Most of the sampling was done in 1993, except for the upland corridors, which were sampled in 1995. Within this study design, it was not possible to test for possible annual fluctuations in abundances that may have affected the results, and this potential bias remained unknown. The analysis assumed that the factor effects were probably stronger than the effects of annual fluctuations.

The experimental design allowed for pairing only of upland and riparian control and clearcut sites. To extend the ANOVA to corridors, all sites had to be treated as independent samples, since there was no basis to pair the corridor sites.

Effects of corridor width and level of partial harvest — Additional sampling was conducted in five partially harvested streamside corridors, one partially harvested upland corridor, and a helicopter-logged streamside clearcut. Partially harvested sites are generally not visible on air photos, so some sites were identified by the forest licensees in the area, and some simply by chance in doing reconnaissance-level sampling for other parts of the project. Three of the partially cut reserve areas had been cut at low levels (25-33% of trees >10 cm dbh removed), while one had been cut at a higher level (74% of trees >10 cm dbh removed). One other sampled riparian forest corridor had been opened by extensive windthrow (48%). Another riparian streamside site had been clearcut salvage-harvested by helicopter after extensive windthrow, but was different from most other clearcut sites in that understorey trees and shrubs remained largely intact, even after clearcutting. Only one upland forested corridor that had been partially harvested (52% of stems >10 cm dbh) was sampled.

Additional analysis used as partial internal — The paired upland-streamside transect design had been used in 1993 for total reserve riparian forest corridors and partially harvested riparian forest corridors. However, after the fact, it was realized that the upland sites were all clearcuts, and the data could not be used within the two-factor ANOVA. (That is why additional upland corridors had to be sampled in 1995 for comparison to the riparian corridors sampled in 1993.) The upland clearcut samples that were not used in the clearcut and corridor to control comparison (N=9) were compared to the five upland clearcut replicates to assess the reliability of the sampling in upland clearcut habitats.

4.3.2 Field sampling

Small mammals were sampled using transects of 25 stations set at 10-m intervals. The small mammal community sampled here was limited to those trappable with Victor mousetraps. This included all mice, voles, and shrews that were present. The only sciurid that was trappable was the Yellow-pine Chipmunk (*Tamias amoenus*). At each station, two Victor mouse snap-traps with wooden treadles or metal treadles modified for increased sensitivity were set for two nights. Total trap-nights were 100 per transect. Riparian transects were usually placed within 5 m of the stream bank, and always within 20 m. At least five traps in every transect were placed on the stream bank, or under the stream bank to attempt to sample small mammals that use the stream itself (Water Shrews and Water Voles). Upland transects were sampled in a straight line parallel to the general stream direction, or in a straight line along the centre of a corridor.

Most sampling occurred between 18 August and 10 September 1993. Additional sampling of the five upland corridors, two streamside, and one upland sites within the partial harvest investigation was conducted between 21 and 26 August 1995. CWD volumes were estimated using a 90-m CWD intercept line along the centre of the transect using the statistical methodology of Lofroth (1992). CWD was defined as any downed log >7.5 cm in diameter at the point of intercept. All trapped small mammals were assigned field numbers, frozen, and later positively identified. Voucher specimens have been deposited in the Royal British Columbia Museum, Victoria, B.C.

4.3.3 Analyses

Analyses were done for each species individually and for two community macroparameters: total abundance of small mammals per transect, and species richness (S, or the number of species captured per transect). In the two-factor ANOVA (clearcut and corridor comparison to controls), any significant interaction effect between the topographic position and timber-harvest treatment factors meant that individual cell means were compared within the factors using one-way ANOVA. Probability of a Type I error (a) was reduced proportionally to the numbers of tests made within factors as suggested by Maxwell and Delaney (1990).

Correlations of CWD volume with small mammal abundance, species richness, and the abundance of the three most common species were attempted. There were insufficient replicates of any single class of corridor width or level of partial harvest to conduct statistical tests, so the data is simply presented and examined. One-factor ANOVA was used to compare the nine additional upland clearcut samples to the first five upland clearcut samples used in the clearcut and corridor to control comparison.

4.4 Results

4.4.1 Effects of topographic position and timber harvesting

Both species richness and total abundance of small mammals showed significant interaction effects between topographic position and timber-harvest treatment, so that individual treatment effects had to be considered within each larger factor (see Appendix H). Species richness of riparian corridors was significantly higher than that of upland corridors (Table 13). There were no significant differences in species richness among any other combinations of topographic position and timber harvesting. Total abundance of small mammals was significantly higher in riparian corridors than upland corridors, and significantly higher in upland controls than both upland corridors or clearcuts.

Although species richness did not differ significantly among controls or clearcuts, species composition did change from controls to clearcuts. The abundance of two species were significantly affected by timber harvesting, but unaffected by topographic position. Red-backed Voles (*Clethrionomys gapperi*) were abundant in controls, less abundant in corridors, and virtually absent from clearcuts. Yellow-pine Chipmunks were also unaffected by topographic position, but were significantly more abundant in clearcuts than controls (where they were absent), and were at intermediate abundance in corridors.

The abundance of two species was significantly affected by topographic position, but unaffected by timber-harvest treatment. Vagrant Shrews (*Sorex vagrans*) and Long-tailed Voles (*Microtus longicaudus*) were significantly more abundant in riparian sites of all timber-harvest treatments than in upland sites.

No significant effects of topographic position or timber-harvest treatment could be shown within this study for Water Shrews (*Sorex palustris*) even though they completely depend on streamside or riparian areas. They were not found in either clearcut or control sites, but were found in streamside forest corridors, even with light partial harvesting.

The abundance of two species was significantly affected by both topographic position and timber-harvest treatment. Deer Mice (*Peromyscus maniculatus*) were significantly more abundant in upland sites than riparian, and significantly more abundant in forest corridors than control forests. Meadow Voles (*Microtus pennsylvanicus*) were significantly more abundant in riparian sites than upland sites, and significantly more abundant in clearcut than control forests. Meadow Voles were most abundant in riparian clearcuts, but this could not be shown significantly within this study because of high variability (see Appendix H). Although the mean abundance of Meadow Voles was highest in riparian clearcuts, this could not be shown to be significantly higher than in other treatments because of high variability (see Appendix H). They were extremely abundant at three of five riparian clearcuts (7-18 captures), but almost absent (only one capture) at two of the five riparian clearcuts.

4.4.2 Effects of coarse woody debris

Coarse woody debris volumes within any of the factors, treatments, or overall were not significantly correlated with total abundance of small mammals, with species richness, or with abundance of the three most common small mammal species, with only one exception. Total small mammal abundance was correlated with CWD volumes within upland sites (r^2=0.38; P=0.01). However, the overall design for examining CWD volume correlation was weak, since only two of the 15 sites had CWD volumes >100 m^3/ha, and these levered the correlation. When 10 additional upland transects (nine from clearcuts, one from a partially harvested corridor) were included in the regression, the strength and significance of the correlation disappeared (r^2=0.06, P=0.24). CWD levels appeared to have a minimal effect on small mammal numbers compared to the strength of the other factors studied here, so that they should not have confounded the analysis of timber-harvest and topographic-position factors.

Table 13. **Significant differences in small mammal abundance by topographic position and timber-harvest treatment. (See Appendix H for full statistics.)**

Community parameters	Treatments (mean number caught/100 trap-night [TN])
CELL MEANS SIGNIFICANTLY DIFFERENT ALONG ROWS OR COLUMNS WHERE INTERACTION EFFECT SIGNIFICANT	
Total abundance	Riparian corridor (20.2) > upland corridor (4.4) Upland control (13.2) > upland corridor (4.4) and upland clearcut (6.2)
Species richness (S)	Riparian corridor (5.4 sp./100 TN) > upland corridor (2.0 sp./100 TN)
ROW OR COLUMN MEANS SIGNIFICANTLY DIFFERENT WHERE FACTOR EFFECT SIGNIFICANT	
Species	
Deer Mouse	Upland (1.5) > riparian (0.4) Corridor (1.2) > control (0), but clearcut (0.2) = both control and corridor
Long-tailed Vole	Riparian (1.5) > upland (0)
Meadow Vole	Riparian (1.5) > upland (0.2) Clearcut (4.0) > control (0.1), but corridor (0.7) = both control and clearcut
Red-backed Vole	Control (10.3) > corridors (4.5) > clearcuts (0.4)
Vagrant Shrew	Riparian (3.8) > upland (0.7)
Yellow-pine Chipmunk	Clearcut (1.0) > control (0), but corridor (0.4) = both control and clearcut

4.4.3 Effects of corridor width and partial harvesting

Forested corridor width appeared to affect small mammal community composition. Each of the four species that were found to be affected by timber-harvest treatment also appeared to be affected by forest corridor width (Table 14). Not enough replicates were sampled within any single class of corridor widths to test these results statistically, but the trends were: Red-backed Voles decreased in number with decreasing corridor width; Deer Mice, Yellow-pine Chipmunks, and Meadow Voles increased in number with decreasing corridor width. There appeared to be no single "threshold" width that determined whether the species composition was more like an unharvested forest or like a clearcut. Rather, individual species appeared to respond to corridor widths independently. This occurred in corridors as wide as 300 m for Deer Mice, or narrower than 100 m for Meadow Voles and Yellow-pine Chipmunks.

Partial harvesting within the ranges considered here (23-72% stem removal) may have little effect on Red-backed Vole abundance. Red-backed Voles were as abundant in control forests (10.3/100 trap-nights, Table 13) as they were in totally reserved corridors wider than 100 m (8.7/100 trap nights, Table 14), and in corridors wider than 100 m with light partial

Table 14. **Number of four small mammals sensitive to timber-harvest treatment on forest corridor transects of different corridor widths.**

	No. caught /100 trap-nights (TN)									
	Streamside forest corridors					Upland forest corridors				
Forested corridor width (m)	500	300	200	63	62	188	150	100	48	39
Meadow Vole	0	0	0	6	1	0	0	0	0	0
Yellow-pine Chipmunk	0	0	0	1	1	0	1	0	0	1
Deer Mouse	0	2	2	1	0	1	3	3	3	1
Red-backed Vole	11	14	1	3	7	2	0	2	5	0

Table 15. Number of small mammals caught in partially harvested streamside and upland forested corridors.

	No. caught /100 trap-nights (TN)						
	Riparian streamside sites						Upland
Forested corridor width (m	200	121	104	54	51	0	45
% partial harvest	32	23	25	48	72	100[1]	52
Common Shrew	0	0	1	0	0	0	0
Deer Mouse	0	2	0	1	1	0	5
Long-tailed Vole	0	7	0	0	1	1	0
Meadow Jumping Mouse	0	1	0	0	0	0	0
Meadow Vole	1	0	2	0	1	0	0
Red-backed Vole	9	11	18	5	3	0	0
Vagrant Shrew	0	4	3	0	0	1	0
Water Shrew	0	2	0	0	0	0	0
Yellow-pine Chipmunk	1	2	1	0	0	0	3

[1] Helicopter-logged clearcut

harvesting (<33% removal, 10.0/100 trap-nights, Table 15). In partially harvested corridors narrower than 100 m, abundance of Red-backed Voles (4.0/100 trap-nights, Table 15) was lower than in control forests, but similar to totally reserved corridors of the same widths (5.0/100 trap-nights, Table 14). Red-backed Vole abundance did appear to be tied in to canopy cover to some extent because when the canopy cover was completely removed by helicopter logging, but almost all understorey trees and shrubs were retained, Red-backed Voles were absent.

For species other than Red-backed Voles, the effects of partial harvesting were not as conclusive because of lack of replication and the general uncommonness of other species in corridors. A possible effect was seen for Yellow-pine Chipmunks and Meadow Voles, which may have been present in streamside forest corridors wider than 100 m perhaps because of the canopy openings created by partial harvesting (Table 15).

4.4.4 Validation of sampling in upland clearcuts

There were no significant differences in the abundance of any species, total abundance, species richness, or CWD volumes between the two sets of upland clearcut samples (Table 16). Therefore, using five replicates for each treatment appeared to provide a consistent estimate of the small mammal communities that was not significantly changed by further sampling.

4.5 Discussion

4.5.1 Riparian species

In general, the habitat associations of small mammals in this study were similar to those found in other studies. Four species were found to be associated primarily with streamside sites. For all but the Water Shrew, the association was statistically significant. For Water Shrews, the association with riparian sites is absolute, but could not be shown to be statistically significant because they were simply too uncommon. Any potential effects of clearcutting on this species still need to be determined.

For the other three species associated with riparian areas, Vagrant Shrews are typically more abundant in riparian areas than in upland areas in the drier parts of their geographical range (Hennings and Hoffmann 1977). This was probably the case in the montane forests studied here. Meadow Vole abundance was probably related to the extent or density of the grass layer, since the grass layer was more dense in clearcut streamside sites than in clearcut upland sites (Vaartnou 1994). However, why Meadow Voles may have been very abundant on some riparian clearcuts and very uncommon on others is unclear. There were no obvious differences in vegetative cover between the sites. Long-tailed Voles appear to prefer habitats with dense shrub cover (Van Horne 1982a). Highest shrub densities were found in streamside areas, and this was not affected by timber-harvest treatment (Vaartnou 1994). This may explain

Table 16. Number of small mammals and species richness on the five upland clearcut samples used in two-factor ANOVA (Table 13) compared to nine additional samples in upland clearcuts.

	No. caught /100 trap-nights (TN)						
	First 5 samples		Additional 9 samples				
Species	Mean	S.E.	Mean	S.E.	$F_{(1,11)}$	P (α)	Effect size[1] (f)
Common Shrew	0.2	0.2	0.2	0.1	0.01	0.93	0.03
Deer Mouse	2.2	1.2	2.7	1.1	0.07	0.80	0.07
Dusky Shrew	0	-	0.1	0.1	0.54	0.48	0.20
Heather Vole	0.2	0.2	0.1	0.1	0.18	0.68	0.11
Long-tailed Vole	0	-	1.0	0.5	2.14	0.17	0.39
Meadow Vole	0.6	0.4	1.9	0.8	1.45	0.25	0.32
Red-backed Vole	0.2	0.2	0.2	0.1	0.01	0.92	0.03
Vagrant Shrew	1.4	0.7	1.1	0.4	0.16	0.70	0.11
Yellow-pine Chipmunk	1.4	0.6	1.4	0.5	0.00	0.96	0.0
Total abundance	6.2	2.0	8.9	1.7	0.98	0.34	0.26
Species richness	3.2	0.4	3.9	0.5	0.77	0.40	0.23

[1] Effect size (f) of Cohen (1988), where 0.4 denotes large effect; 0.25 medium effect; and 0.1 small effect.

why Long-tailed Vole abundance was not significantly affected by timber-harvest treatment in this study.

4.5.2 Effects of timber-harvest treatment

Riparian forest corridors supported the highest small mammal abundance and the greatest variety of species in this study. The corridors supported species found in upland clearcuts (Deer Mice and chipmunks), riparian clearcuts (Meadow Voles and chipmunks), and uncut forests (Red-backed Voles). These corridors were the only treatment in this study in which all the small mammal species occurred. This cannot be entirely attributed to edge effects because upland corridors had the same amount of clearcut-forest edge habitat as riparian corridors, but did not even approach the species richness or small mammal abundance of riparian corridors. The juxtaposition of clearcut and forest habitats in forest corridors, in conjunction with the riparian influence on vegetative composition (and perhaps other factors not directly measured), probably yielded the highest small mammal diversity possible in this area.

The Red-backed Vole was the only species significantly associated with forests within this study. It appeared that forest corridors of 100 m wide provided suitable habitat for this species, but that abundance declined in corridors narrower than 100 m. This may have been because density declined near forest edges, as found for Western Red-backed Voles (*Clethrionomys californicus*; Mills 1995). However, the transect sampling method used does not measure density because the sampled lateral width is unknown. If the effective sampling width was >50 m, then relative abundance would have declined in corridors narrower than 100 m, even though density may have been constant. True density could only be determined by finding the effective sampling area using live trapping grids and outlying sampling transects (e.g., Van Horne 1982b).

4.5.3 Uncommon species

The Common Shrew was uncommon within this study, but was generally limited to forested or partially harvested sites. Within clearcuts, they were only found where regenerating conifers were at least 1.5 m tall. Other studies found Common Shrews to decrease dramatically in numbers after clearcutting (Sims and Buckner 1973), although Martell (1983, 1984) found the response to clearcutting to be variable. This may depend on the amount of residual vegetation remaining after clearcutting.

Three other species were very uncommon, and no habitat-use patterns could be shown within this study for these species (see Appendix H). Only 14 Heather Voles (*Phenacomys intermedius*) were caught, 11 of which were in riparian forests, i.e., both controls and corridors. The habitat affinities of Heather Voles are poorly known because they are uncommon anywhere and occur in a variety of habitats (Innes and Millar 1982). Only three Dusky Shrews (*Sorex monticolus*) and two Meadow Jumping Mice (*Zapus hudsonius*) were captured within this study.

4.5.4 Riparian and upland small mammal communities

Other published studies directly comparing forested streamside to upland small mammal communities with a paired-transect study design produced different results depending on the vegetative type (and the resultant small mammal communities) studied. In western Oregon, there were small mammal community differences between riparian and upland sites in mature Douglas-fir stands where the habitat differences between riparian and upland sites were pronounced (McComb et al. 1993a; Doyle 1990). In those studies, riparian sites had lower overall canopy closures, with a higher deciduous component and lower coniferous component, and very pronounced differences in understorey plant species and coverage. However, when the riparian and upland habitats were very similar and were both red alder stands, no differences in small mammal communities were found (McComb et al. 1993a). In the Okanagan Highlands, both vegetation and small mammal communities differed between riparian and upland sites.

In the Coast Range of Oregon, McComb et al. (1993a) found species composition differed between upland and riparian forests, although species richness and abundance did not differ. In the Cascade Range of Oregon, Doyle (1990) found greater species richness and abundance in riparian forests as well as a change in species composition. In the Okanagan Highlands, species composition was different in riparian and upland forests because four species were more common in riparian forests than upland forests. However, the higher abundance and species richness in riparian forests were masked by high variability, so that no significant difference could be shown.

McComb et al. (1993a) suggested that both upland and riparian areas must be included in forest reserve networks, because of the species differences between upland and riparian forests in western Oregon. In the Okanagan Highlands, riparian forest corridors contained all small mammal forest species present in the Okanagan Highlands, suggesting that priority should be given to maintaining riparian forest reserves over upland forest reserves in the Okanagan Highlands. The only species more common in upland areas, the Deer Mouse, was also abundant in clearcuts, so that its conservation may not depend on the retention of forest corridors.

However, the studies cited above from Oregon only considered unharvested forests. These results cannot be assumed to apply to forest corridors — particularly at widths of <100 m where habitat suitability may decline, as was the most likely case here for Red-backed Voles. Further assessments of the habitat suitability of forest corridors is required.

4.6 Implications for Timber-Harvest Management

- Clearcut timber harvesting changed the species composition of small mammal communities, but only reduced total abundance in upland areas.
- No small mammal species was limited to upland areas, while three species were significantly more abundant in riparian areas and one other was limited to riparian areas. Forested corridors along streams supported all of the mouse, vole, and shrew species found in the Okanagan Highlands.
- Upland forest corridors supported lower diversity and abundance of small mammals than did riparian forest corridors.
- Only Deer Mice were most abundant in upland areas. However, since they were also abundant in clearcuts, their maintenance within forest corridors need not be a priority.
- Optimal corridor widths could not be defined within this study, although edge effects in species composition started to be shown in corridors as wide as 300 m. The most common forest species (Red-backed Vole) was present in all forest corridors, but most abundant in corridors wider than 100 m.
- Partial harvesting in forest corridors is acceptable since levels below 33% did not appear to affect small mammal communities. Higher levels of removal may have affected small mammal communities but could not be definitively separated from possible width effects in this study.

The following management recommendations are made for the Montane Spruce Zone:

- In the connection of forests across a landscape, forest corridors wider than 100 m should be reserved along streams in preference to upland corridors to maintain small mammal diversity.
- Partial cutting in the forested reserve corridor is acceptable, for forest-dwelling small mammals, but is preferred at low removal levels (<33%) until the effects of higher removal levels are confirmed.

5 AMPHIBIANS

5.1 Introduction

Amphibian populations, in general, are of concern because of noted declines world wide (Barinaga 1990; Blaustein et al. 1994a). Populations in agricultural and urban areas have been affected by isolation of habitats by roadways and railways (Reh and Seitz 1990), and by isolation of breeding ponds from forested areas (Laan and Verboom 1990). Pond-breeding amphibians depend on their breeding ponds and on access to them. The effects of clearcutting on forest-floor-breeding Plethodontid salamanders are devastating (Petranka et al. 1993), but the effects on pond-breeding amphibian populations are poorly known.

Range maps in Green and Campbell (1984) indicate that four species of amphibians were most likely found in the Okanagan Highlands: Western Toad (*Bufo boreas*), Spotted Frog (*Rana pretiosa*), Pacific Treefrog (*Hyla regilla*), and Long-toed Salamander (*Ambystoma macrodactylum*). Additional species found in the Okanagan valley bottom include Great Basin Spadefoot Toad (*Scaphiopus intermontanus*) and Tiger Salamander (*Ambystoma tigrinum*). The Northern Leopard Frog (*Rana pipiens*) has occurred in the Okanagan valley bottom, but has not been recently found there. The main range of the Wood Frog (*Rana sylvatica*) is apparently north of the study area, although scattered observations south of the study area indicated that it may have been found there. All these amphibians depend on ponds or stream reaches with still water for breeding. After breeding, only the adult Spotted Frogs use riparian areas extensively; the adults of other species may wander very widely and their non-breeding habitat preferences are poorly known (Green and Campbell 1984). However, if breeding occurs, as determined by presence of eggs or larvae, then there must be adult populations in the surrounding area, although the actual extent of that area may remain unknown.

The presence of eggs or larvae at a breeding site would not by itself indicate whether the site was optimal for breeding or was a reproductive sink. Breeding is successful only when the larvae survive to metamorphosis. Similarly, if the young adults do not survive to breeding age, then reproduction is also unsuccessful. Estimating the survival of amphibian adults after they metamorphose and leave the breeding ponds is not simple, and was not judged within this study, although we did attempt to estimate whether larvae did survive to metamorphosis at the selected sample sites.

This study began in the MSdm1 Zone of the Okanagan Highlands in 1993, where it continued through 1995. In 1995, the scope was extended to the ESSFxc and ESSFdc1. In 1995, it also became obvious that just presence/absence of larvae would not even begin to describe the potential complexities of amphibian breeding-site choice and survival. Consequently, extra measurements were made of temperatures at a few key sites, and a few breeding sites were checked weekly for larvae to follow their progress through the summer. It was hoped that these observations might provide insight to refine future studies.

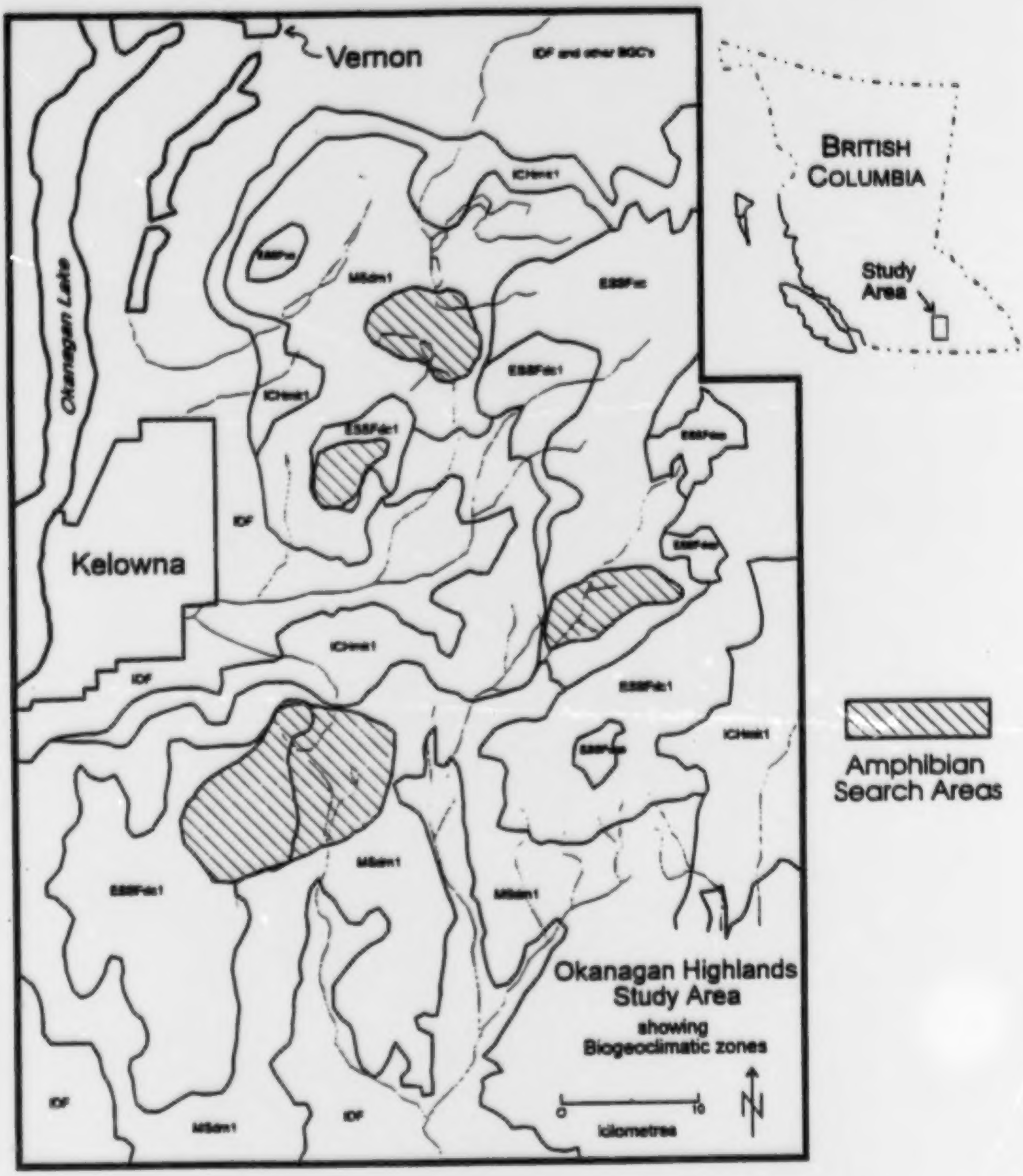
The specific objectives of this amphibian study were to:

- Determine the species of amphibians breeding in the Okanagan Highlands.
- Determine whether these amphibians would breed in wetlands or water bodies within clearcuts, whether larvae could survive to metamorphosis within clearcuts, and if there was any relationship between the distance from the forest or the time since clearcutting, and the wetlands or water bodies used for breeding.
- If possible, develop recommendations for timber-harvesting guidelines that would retain the diversity of amphibians in riparian areas of the Okanagan Highlands.

5.2 Study Area

The study area was in the Okanagan Highlands east of Kelowna, B.C. (Figure 10). The northern part of the study area was in the Vernon Forest District and the southern part was in the Penticton Forest District, both in the Okanagan Timber Supply Area of the Kamloops Forest Region of the B.C. Ministry of Forests.

This study focused on the two predominant biogeoclimatic zones of the Okanagan Highlands. The Okanagan Dry Mild variant of the Montane Spruce Zone (MSdm1) predominated at medium elevations (1300-1600 m), while the Very Dry Cold Engelmann Spruce–Subalpine Fir subzone (ESSFxc) and the Okanagan Dry Cold Engelmann Spruce–Subalpine Fir variant (ESSFdc1) of the Engelmann Spruce–Subalpine Fir Zone predominated at higher elevations (1600-2000 m) (Lloyd et al. 1990). The forest vegetation of the MSdm1 was dominated by a fire-caused disclimax of lodgepole pine in upland stands, which generally proceeds to a spruce (white-Engelmann hybrid) and subalpine fir climax in the absence of stand-replacing fires. Spruce and subalpine fir were common at all stages of regeneration along


Vernon
IDF and other BGC's
ICHmk1
MSdm1
ESSFdc1
ICHmk1
ESSFdc1
Okanagan Lake
IDF
Kelowna
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IDF
ESSFdc1
ICHmk1
MSdm1
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MSdm1
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Okanagan Highlands
Study Area
showing
Biogeoclimatic zones
0
10
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BRITISH
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Study
Area
Amphibian
Search Areas

Figure 10. Amphibian search areas in the Okanagan Highlands.

creeks. The ESSFxc and ESSFdc1 have been grouped here and collectively called the ESSF. The upland and riparian forests of the ESSF were dominated by Engelmann spruce and subalpine fir, although lodgepole pine was abundant after stand-replacing fires.

The study area was chosen for two reasons. First, there were extensive existing clearcuts, and a high priority to harvest much of the remaining forest because of the prevalence of older lodgepole pine stands susceptible to Mountain Pine Beetle attack. Second, the area is a plateau with numerous riparian sites, including many low-gradient streams and many small ponds and wetlands that could make ideal amphibian breeding sites. Wetlands are generally dominated by moss, sedge, or willow. Cattail marshes are of limited occurrence in the study area. Local climate information was also available from the station at McCulloch (49°48'N, 119°12'W, elev. 1250 m, Environment Canada 1993 and unpublished data), which is within the study area.

5.3 Methods

This study began in the spring of 1993 without a good idea of what sort of water bodies, ponds, or wetlands would be used for breeding by amphibians in the Okanagan Highlands. A selection of 15 ponds and wetlands within forest cover and 31 within clearcuts were marked on 1:15 000 colour air photos of two areas within the MSdm1 of the study area (see Figure 10). The definition of a potential amphibian breeding site quickly became apparent based on differences between sites, and the actual spots within wetlands that were used for egg laying. Any water body, or portion of a water body, with a water depth of at least 20 cm during the May breeding period had the potential to be used by amphibians for breeding. The single site that had <20 cm of water was therefore eliminated from the study, leaving 15 forest sites and 30 clearcut sites in the MSdm1. Distance to forest and years since clearcutting were noted for every clearcut site. MSdm1 sites were visited once in June 1993, once in June and once in early August 1994, and once in June and once in late July 1995. Only sites where larvae were found were visited again in a single year to monitor the development of larvae toward metamorphosis and larval survival.

After looking at the results from the first two years, it became apparent that the MSdm1 sites had not been chosen entirely without bias. The very small (<0.04 ha) wetlands that were often used for breeding within clearcuts in the MSdm1 were not visible on air photos under forest cover. These small sites are easily recognized on colour air photos within clearcuts, because of the bright green vegetation surrounding them. However, the canopy cover was fairly complete over such small sites within forests, and they could not be reliably detected. Thus, on average, larger wetlands were chosen in forests than in clearcuts, and the results were not strictly comparable because of this bias. In 1995, sites were chosen for examination in the ESSF and an attempt was made to remove this site-choice bias. The only wetlands and ponds chosen for examination within clearcuts were sites that probably would also have been distinguishable within forests on air photos. In the ESSF, 26 sites were chosen within unharvested forest, and 30 sites within clearcuts at various distances from forest cover. ESSF sites were visited once in late June and again in late July 1995. The only sites checked a second time in the ESSF were sites that were likely too cold for amphibian breeding during the first check (<6°C), and sites where larvae were found on the first check to describe the development of larvae towards metamorphosis and to investigate larval survival.

Ponds or wetlands chosen as study sites were searched until most of the shallow open water had been covered at a site. Most sites were <2 ha, and average time spent searching during daylight at each site was 30 to 60 minutes the first time they were visited. Later searches of the same sites were made more quickly because most breeding locations were known within larger riparian sites. On return visits to sites, these specific breeding places could be visited immediately to determine if breeding had taken place. If no larvae were found at that spot, or fewer species of larvae than previously, then the search for larvae was again widened within the site.

Only larvae of the Western Toad, Spotted Frog, Pacific Treefrog, and Long-toed Salamander were found in the Okanagan Highlands. Adults of the first three species were also commonly seen, although only one adult Long-toed Salamander was ever found, despite turning over a few logs at every site visited. The larvae of each species was found to be sufficiently different that they could be reliably identified at a 1-m viewing distance. Therefore, they did not have to be caught to be identified. Approximate numbers of larvae of each species were noted, but since neither search area nor time of search were equal across sites, no estimates of larval densities were possible. In addition, larvae numbers were extremely difficult to estimate in algae-filled fens or marshes and in some of the coniferous swamps where tannins reduced visibility to less than 15 cm. Therefore, analyses only considered the presence or absence for each species at each site.

After the first two seasons of pond and wetland checks for amphibians in the MSdm1, it seemed that temperature was playing a key role in both initial site choice of adults for egg-laying, and the rate at which temporary ponds used for egg-laying would dry up in the summer. Temperature may be especially important to amphibians, since the rate of amphibian larval development is directly correlated with higher water temperatures (Smith-Gill and Berven 1979; Berven 1982). During 1995, additional observations were made to provide some insight into the relationship of temperatures and amphibian breeding, and to provide reconnaissance-level data that might guide future studies. Observations of breeding activity were noted, and temperatures were measured on ponds in May, while field staff were laying out breeding bird plots. In addition, six of the MSdm1 ponds were checked weekly from late June through July to determine when larvae leave the ponds, and how reliable any estimate of larval survival might be that is based on two visits to a breeding pond in a summer.

Not all larval species were equally detectable. Early in the breeding season, frog and toad larvae were usually present at the surface, and presence could be detected by the roiling of the surface as tadpoles reacted to the observer's approach. To detect salamander larvae, which usually lay at the bottom of the ponds, a dip net was run through the pools where the bottom could not be seen. Later in the season, older Spotted Frog larvae were usually quite wary and dove deep into algal mats upon sensing the observers, so that they were often difficult to find later in the breeding season. Western Toad larvae, when present, usually numbered in the thousands. Long-toed Salamander larvae were only rarely found in numbers of more than 50. The search time chosen was a trade-off between easily detected and more difficult to detect species. Searches for Western Toads can be completed quickly, but more thorough searches must be made for salamander larvae. In addition, Long-toed Salamanders can take up to three years to mature in high-elevation ponds (Howard and Wallace 1985), and the very small first-year larvae can be difficult to detect. Detailed notes were not made on larval size of Long-toed Salamanders, but there did appear to be at least two cohorts of size-classes, so that overwintering of larvae probably did take place. For these reasons, analyses were only done within species, rather than across species.

Contingency tables and the χ^2 statistic were used on presence or absence data for the ESSF, where an attempt was made to eliminate site-choice bias between forests and clearcuts to test for significant differences within species between forested and clearcut sites. Other analyses were only completed within clearcut or forest habitats because of the potential site-choice bias. Logistic regression (where y-values are either 0 or 1 for absence or presence, respectively) using the Newton-Raphson iterative technique was used to determine if time since timber harvest and distances from forest cover were correlated with presence or absence for each species.

5.4 Results

5.4.1 Habitat relationships

Larvae of the Western Toad, Spotted Frog, Pacific Treefrog, and Long-toed Salamander were found in both the MSdm1 and ESSF. Larvae were found at 32 of the 45 MSdm1 sites searched in 1993, 37 of the sites in 1994, and 36 of the sites in 1995. Considering all three years together, larvae were found at 41 of the 45 MSmd1 sites. Fewer sites (21 of 56) contained larvae in the ESSF. Every possible combination of the four species of larvae were found at sites, including all four species at four of the sites.

Within the searched areas, amphibian larvae were found in a variety of natural sites, as well as of sites that were modified or created during timber harvesting that were at least 20 cm deep. These disturbed sites included ditches beside active roads, puddles in old landing sites, puddles in deep machine-caused ruts in clearcuts, ponds in gravel-extraction sites, puddles adjacent to excavator-mounds that are used as seedling planting sites, and puddles on "winter-logging" roads where no roadbed was ever constructed and the deep ruts fill with water. The smallest "breeding pools" (isolated from surrounding water) that were used were the puddles beside excavator mounds that had surface areas of approximately 1 m^2. Both Spotted Frog and Pacific Treefrog larvae were sometimes found in these sites. These pools beside mounds were found within larger wetland sites where there were other breeding pools. Natural sites used for breeding included coniferous- and mixed-tree swamps, shrub swamps, sedge marshes, moss-dominated fens, quaking bogs, and a variety of beaver ponds, marshes, and swamps. No amphibian larvae were found in the puddles formed in the rootball holes beside large blown-over trees with tipped-up roots. At least a few of these puddles were found in every riparian site that had large conifers.

In both the MSdm1 and ESSF, larvae of all four species were found both within clearcuts and within forests (Table 17). Within the ESSF, each species was

Table 17. **Number of ponds (minimum 20-cm deep in June) in forests and in clearcuts where amphibian larvae were present in the MSdm1 and the ESSF.**

		Number of sites where larvae were found								
		MSdm1						ESSF		χ^2 for ESSF forest to clearcut comparison
		1993		1994		1995		1995		
		Forest	Clearcut	Forest	Clearcut	Forest	Clearcut	Forest	Clearcut	
Species	(N sites)	(15)	(30)	(15)	(30)	(15)	(30)	(26)	(30)	
Spotted Frog		4	14	5	20	8	15	5	13	3.71
Pacific Treefrog		4	18	5	20	8	19	2	8	3.42
Western Toad		3	5	3	7	5	6	3	7	1.32
Long-toed Salamander		4	5	4	13	5	13	1	5	2.39

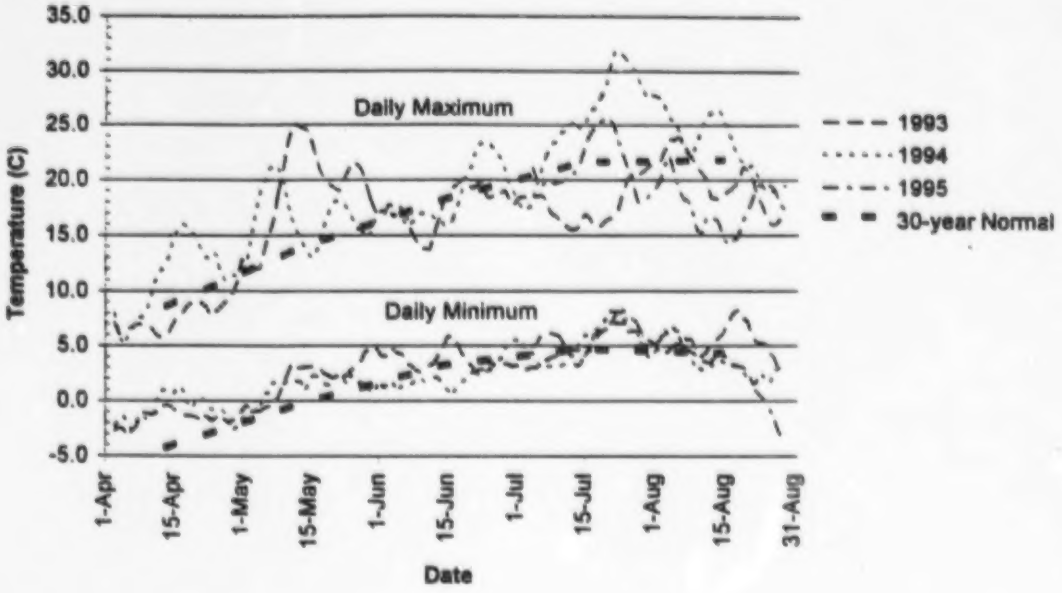
found at a higher proportion of sites within the clearcuts than within forests; no difference was significant at α=0.05. No statistical comparisons were made within the MSdm1 because of the site-choice bias previously noted.

Neither the number of years since clearcutting nor the distance to forest had a significant correlation with amphibian presence or absence at breeding sites in clearcuts for any species. Logistic regression r^2 values were between 0.001 and 0.10 for all attempted correlations, and no values were significant. Clearcut ages examined varied from 3 to 24 years. Larvae of each of the four species were found at least once at sites over 1 km from the nearest forest in the MSdm1. In the ESSF, the site furthest from a forest was only 350 m distant. Clearcuts in the ESSF tended to be smaller than in the MSdm1 because the MSdm1 has been subject to extensive Mountain Pine Beetle salvage and control logging, so that some clearcuts were very large (over 1000 ha).

5.4.2 Larval survival

An attempt was made to follow breeding success within individual breeding sites by visiting them twice over the summer period in 1994 and 1995 to see if larvae did emerge from the ponds and pools. However, weekly visits to six selected ponds in 1995 showed that this was far too few visits to accurately predict whether larvae had survived to metamorphosis in individual ponds. Based on calling at breeding sites, and first appearances of eggs and larvae, the primary egg-laying period for amphibians in the Okanagan Highlands begins in late April and continues through to mid- or late May. This coincides with the period when average daily minimum temperatures rise above freezing and when average daily maximum temperatures rise above 10°C (Figure 11). The breeding period of Long-toed Salamanders did not appear to be as synchronous as the three species of frogs and toads. Anuran eggs were only found in the first half of May (although some were probably laid in April when observers were not in the field for any aspects of this riparian study), while Long-toed Salamander eggs were (once) found as late as July.

In 1995, larvae were found metamorphosing and leaving ponds as early as the first week of July, although most appeared to leave later in July or in early August. The average time from egg-laying to metamorphosis appeared to be in the range of 90 to 120 days. Because of this range in timing, an original estimate that it might be possible to get an indication of whether larvae survived to metamorphosis by visiting a breeding pond twice in a summer turned out to be far too optimistic. When visiting ponds or puddles that had dried up by late July, it was impossible to accurately estimate if any larvae had left the pond before it had dried up. For example, the first part of July 1995 was dry and warm, but the latter half was wet. One of the shallow (20 cm) ponds that was visited weekly in a clearcut was observed to dry out by mid-July, and the larvae were found dead at the bottom of the pond as it dried. Based on the general level of development, none of the larvae from this pool reached metamorphic climax in 1995. However, when visited one week later, rain had replenished the pond and all sign of amphibian larvae had vanished. At this sort of temporary pool, visits must be at least weekly to develop any sort of estimate of larval survival to metamorphosis.



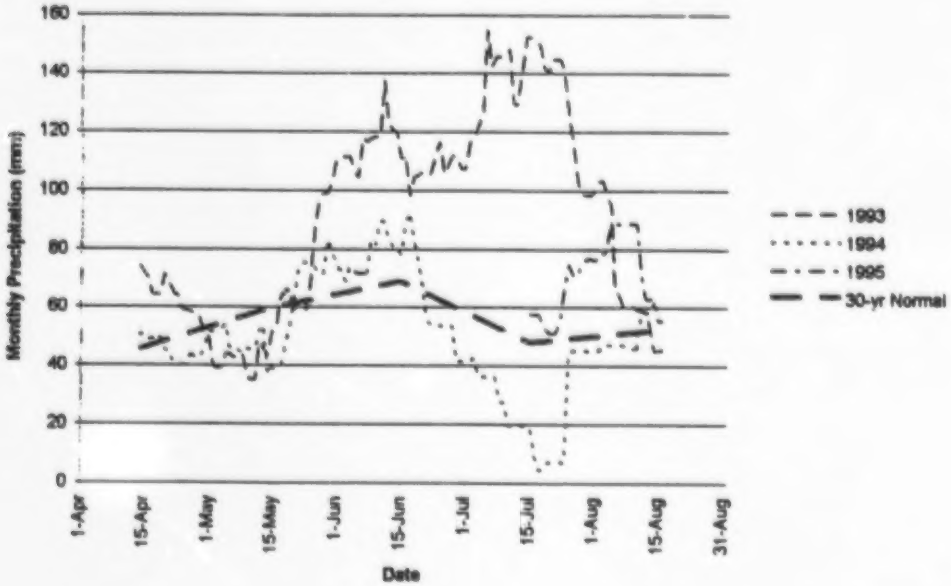


Figure 11. Daily maximum and minimum temperatures (running seven-day averages) compared to monthly normals (top), and 30-day running average precipitation totals compared to monthly normal totals (bottom) for McCulloch for April to August 1993-95.

5.4.3 Role of temperature

Coniferous canopy cover appeared to play a key role in water temperatures and have some influence on adult choice of breeding site. Water temperatures were taken at noon in three small (<0.02 ha) breeding ponds in clearcuts and in three small (<0.02 ha) ponds under coniferous canopies on 3 May 1995. The procedure was repeated on 18 May in three other small ponds in clearcuts and three other small ponds under coniferous canopy. These small ponds under coniferous canopy were the type that simply could not be identified on 1:15 000 air photos, because there was virtually no canopy break over them. On May 3, when ambient air temperatures were 15°C, water temperatures were 12-13°C in small clearcut ponds and only 3-6°C under coniferous canopies. Temperature at the same time in a larger (0.15 ha) pond surrounded by coniferous forest was 11°C. This 0.15 ha pond was clearly visible on the air photo because of a distinct canopy break that allowed the sun to reach the pond. The sun did not reach the 0.02-ha ponds, except as filtered through a coniferous canopy. On 18 May, when ambient air temperatures were 12°C, water temperatures were 12-16°C in the small clearcut ponds, but only 2-6°C in small ponds under coniferous canopies. Since temperatures below 6°C are lethal to the young embryos of Spotted Frogs (Licht 1971), amphibian breeding during May is unlikely in very small 0.02-ha pools under coniferous canopies.

July 1994 was extremely hot and dry, especially when compared to July 1993 and 1995 (Figure 11). In 1994, mean July temperatures were 15% above 30-year normals and rainfall was 60% below 30-year normals. All the temporary pools (in both clearcuts and forest) that were <50 cm deep when first examined in July 1994 were dry, or <2 cm deep, by early August 1994. The only exceptions were sites with inflowing water, where old beaver ponds on small streams had created breeding pools for amphibians. In July 1993, mean temperatures were 14% below the normals and rainfall was 218% above. In 1993, only pools with <20 cm water in June dried out, and they took until September to do so. Consequently, all amphibian larvae raised in the ponds had the potential to leave the ponds as adults in 1993, but not in 1994 unless they left the temporary pools in early to mid-July. In 1995, summer temperatures and precipitation were closer to normal (April to June data was unavailable in 1995), and most of the temporary ponds that were only 20 cm during the breeding period did dry up by late July.

5.5 Discussion

The principal effect of clearcutting on the reproduction of pond-breeding amphibians in the Okanagan Highlands appeared to be the expansion of breeding habitat through enhanced suitability of small (<0.02 ha) pools, which would have otherwise been unsuitable under coniferous canopy cover. The mechanism also appeared to be fairly simple — the small pools under complete coniferous canopy cover were too cold to allow spring breeding, but the same size pools were warm enough in clearcuts. The trade-off appears to be that many of the small pools were temporary, and they did not retain water through dry, or even normal, summers. There is a strong possibility that these small pools in clearcuts are acting as reproductive sinks, i.e., they are very attractive for egg-laying because they are warm in the spring, but have relatively poor larval survival because they dry out so quickly in even normal summers. That many of these sites are made by humans may also be cause for concern — the tiniest sites found for breeding were all associated with timber-harvesting, silvicultural, and road-building activities (e.g., ditches and wheel ruts). These types of sites do not necessarily have any natural equivalent. In the absence of these sites, the amphibians would have bred in larger wetlands and pools, i.e., typically in beaver ponds and pothole wetlands. These sorts of wetlands appear to be more permanent and not as subject to drying out in mid-summer, and therefore possibly more stable sites for larvae to develop to reach adulthood.

There is probably a complex relationship between speed of amphibian larval development and the speed of evaporation in temporary breeding pools. The rate of evaporation in these pools will be correlated with higher temperatures (since high temperatures generally only occur along with lack of rainfall in the study area), and rates of larval development will also be correlated with higher temperatures. However, only by much more frequent and intensive observations than were made in this study could the critical points of temperature, lack of rainfall, and larval success in temporary ponds of different depths be determined.

Variations in frog populations are typically caused by extreme variations in larval survival (Berven 1990), and this appears to be the situation here, too. So, while the number of potential breeding sites might be enhanced within clearcuts, survival of larvae to adulthood was not necessarily enhanced. Since this was only a presence/absence survey, neither actual reproductive potential nor survival could be estimated, and the absolute effects of clearcutting on the population as a whole remain elusive.

For a better understanding of how clearcutting is affecting the amphibian communities, it would also be important to know how clearcutting affects adult survival, and how far adults will migrate to breeding ponds from preferred summertime or hibernating habitats. While some breeding pools may be created, or their egg-laying suitability enhanced by clearcutting, there will only be any benefit to populations if adult survival is not affected. Adult amphibian habitat preferences and the migration distances and fidelity to particular breeding sites are extremely poorly known. In this study, with its limited scope, the assumption was that if larvae are found, adults must be surviving somewhere nearby. Without knowing migration distances, "nearby" is impossible to define. We do not know where adults are during the non-egg-laying period, much less whether survival is better in forests or clearcuts.

We do not yet have the sort of data needed to predict amphibian survival within clearcuts. If small pools near wetlands continue to be very attractive egg-laying sites, but larval survival is low in hot or normal summers, then the effects of the possible warming trend because of increasing atmospheric CO_2 concentrations (Harding and Taylor 1994) may only exacerbate this effect. In addition, increased UV radiation due to thinning of the stratospheric ozone layer may be responsible for declining amphibian populations — including the Western Toad in the western U.S. (Blaustein et al. 1994b), which was one of the species studied here. If UV radiation is lower under forest canopies, then forests may have a large role to play in the survival of amphibian populations.

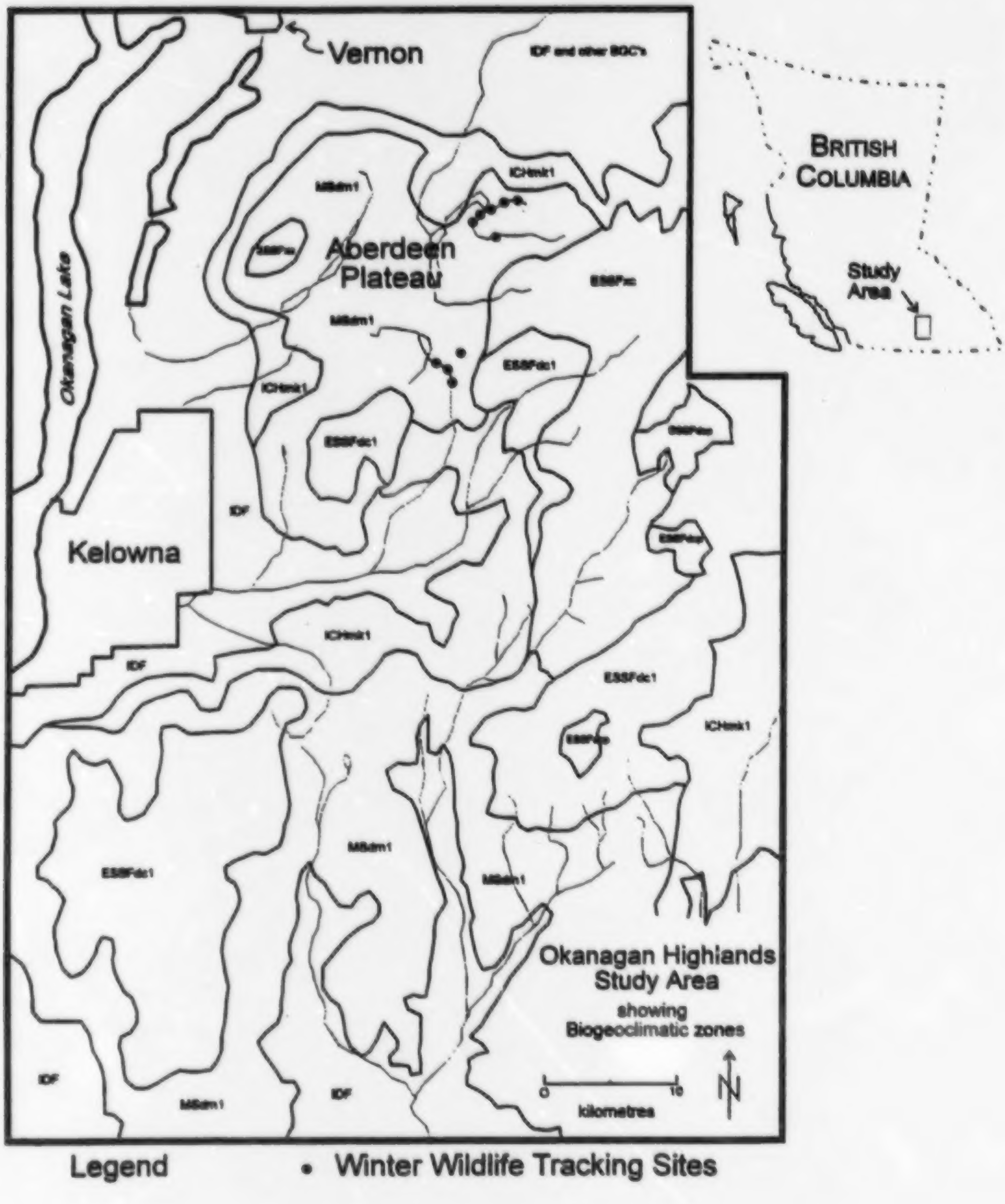
We could not estimate predation rates on larvae within this study and how that might affect survival. These small pools used for breeding in clearcuts may be reproductive sinks, not only because they dry out more quickly than larger wetlands, but because predation rate may be higher in them as well. When we approached breeding pools later in the breeding season, larvae appeared to be extremely aware of the observer's approach and would dive into algal mats or under logs for cover. This sort of cover usually did not exist in the smallest pools (e.g., ditches and tire ruts) used for breeding, so that predation rates are likely to be higher there. We actually observed only one instance of predation when Common Ravens were once observed feeding on Spotted Frog larvae in a roadside ditch. However, Common Garter Snakes are a predator of amphibian larvae and were occasionally observed in the breeding ponds.

The only other instance of predation noted was in one of the ponds that were visited weekly in 1995. The small pool (about 10 × 5 m) was in a gravel pit beside a wetland and was used for breeding by Long-toed Salamanders, Pacific Treefrogs, and Western Toads. In early July, most larvae had developed four legs, were developing adult coloration, and a few were hopping onto the muddy shore. One week later, no amphibian larvae were left in the pond and fresh bird tracks covered the muddy shore (possibly ravens and/or shore birds). The newly metamorphosed adults almost certainly suffered high predation rates in moving from the breeding pool into the main wetland. Any larva born within the adjacent wetland (which would have been the only breeding site before the gravel pit was built) would probably have had a much higher chance of survival, since it would not have had to run any gauntlet between rearing pool and adult habitat.

Woodland salamanders appear to be much more sensitive to forest disturbance than the pond-breeding amphibians found in the Okanagan Highlands (see Petranka et al. 1993). Woodland salamanders do not breed in ponds, but depend on rotting logs on the forest floor for breeding and for adult survival. No woodland salamanders occurred in the Okanagan Highlands where natural forest disturbances from large wildfires are relatively frequent. Woodland salamanders may be examples of species with evolutionary strategies that put them at an advantage in habitats where small-scale disturbances (e.g., tree-fall) are frequent and large stand-replacing fires relatively infrequent.

No species appeared to "exclude" any other species from breeding at a particular site in a given year. Every possible combination of the four species of larvae were found at sites, including all four species at four of the sites. Still, the possibility of competitive relations between these species cannot be ruled out. The exact nature of these larval relationships can be difficult to deduce because the both timing of egg-laying and the adult choice of microsites can be important factors (Petranka et al. 1994), and these were not examined here.

Each of the four species of amphibians found in the Okanagan Highlands used a wide variety of riparian habitats for breeding, with the sole requirement appearing to be standing water at least 20 cm deep during the egg-laying period, and perhaps exposure to sunlight through a break in the coniferous canopy, in either natural wetlands, or small human-made sites near wetlands. However, since all the sites considered had that much water, and not all sites were used for breeding by all species, there remains a considerable amount of natural variation that cannot be accounted for with a simplistic analysis based on


Vernon
IDF and other BGC's
BRITISH COLUMBIA
Study Area
ICHmk1
MSdm1
Aberdeen Plateau
ESSFxc
Okanagan Lake
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ICHmk1
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IDF
Kelowna
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MSdm1
Okanagan Highlands Study Area
showing Biogeoclimatic zones
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Legend
• Winter Wildlife Tracking Sites

Figure 12. Winter wildlife study sites on the Aberdeen Plateau of southern British Columbia.

clearcut or forest habitat that surrounds the breeding ponds. These species of amphibians will breed in wetlands surrounded by clearcuts, but the extent to which they may rely on forest at some stage of their life cycles is still unknown. There may also be threshold effects as the percentage area that remains forested within certain critical distances of breeding ponds declines below specific values. These threshold effects may only become apparent as the final passes are made within the first rotation of timber harvesting. At this point, it would appear that better information on adult survival, and critical seasons or habitats, is needed before definitive conclusions can be made of amphibian population persistence in managed forests.

6 WINTER WILDLIFE

6.1 Introduction

This part of the study addresses winter-active wildlife in riparian systems in the Okanagan Highlands. The specific objectives of the winter wildlife study were to:

- Determine whether winter-active wildlife uses Riparian Management Zones (RMZs) along creeks differently when the upland buffer is clearcut, partially cut, or totally reserved.
- Determine if there is a minimum width of forested RMZ corridor that is useful for different species.
- Make recommendations for timber harvesting adjacent to riparian zones that will protect the wildlife of these zones.

6.2 Study Area

The study area was in the Montane Spruce Zone on the Aberdeen Plateau north-east of Kelowna and south-east of Vernon, B.C. (Figure 12). The northern part of the study area is in the Vernon Forest District and the southern part is in the Penticton Forest District, both in the Okanagan timber supply area of the Kamloops Forest Region. The area was chosen because of the high levels of existing cut and the high priority to harvest much of the remainder because of the presence of large stands of older lodgepole pine, which is highly susceptible to Mountain Pine Beetle attack. There are also many low-gradient streams without pronounced gullies that were chosen as the riparian type for this study. The actual sites were along Belgo Creek and Hilda Creek south of Ideal Lake (Figure 13) and along Curtis Creek and Heart Creek upstream of Aberdeen Lake (Figure 14). Lodgepole pine predominates in young upland stands in the area, which generally proceed to a spruce (white-Engelmann hybrid) and subalpine fir climax in the absence of fire. Spruce and subalpine fir are common at all stages of regeneration along creeks and wetter sites. Study-site elevations ranged from 1265 to 1350 m.

6.3 Study Design and Methods

One objective of this study was to compare the effects of three types of harvest treatments in RMZs clearcut, partial cutting, and total reserve. The study sites and design were dictated by the very few sites that had been partially cut in RMZs along creeks. Clearcutting has been the preferred method of harvest in the area because of the large even-aged lodgepole pine stands. At three sites along Curtis Creek, partial cuts had been made in the RMZ.

The clearcut sites (Heart 1, Heart 3, and Hilda 1) were clearcut to the stream banks with a narrow no-machine buffer (NMB), leaving some non-merchantable timber and some understorey trees and shrubs close to the stream bank. Two sites had been partially cut throughout (Curtis 1 and 3) and two sites had been totally reserved (Belgo 1 and Belgo 2). Curtis 2 had been totally reserved on one side of the creek and partially cut on the opposite side. Mountain Pine Beetle attack and blowdown dictated the extent of the partial cuts. Curtis 1 had been partially cut at levels of 50 to 85% removal (overall mean of 75%), while removal levels were between 25 and 33% at Curtis 1 and 3.

Two additional transects were added that sampled reserves along creeks where the other side of the creek was completely forested (Belgo 3 and Heart 2). These provided some measure of winter use by wildlife in unharvested areas.

At each site, two transects, 100 m apart, were established at right angles to the stream. Each transect started in the clearcut approximately 100 m from the RMZ, continued into the RMZ, across the creek, through the RMZ on the far side of the creek, and approximately 100 m into the next clearcut. The forested sites were sampled to 100 m from the creek bank. Each transect was sampled on foot five times between 2 December 1992 and 3 February 1993. Heart 2 and Heart 3 were sampled three times each because of difficult access in deep, unconsolidated early winter snows. All tracks crossing the transect and birds heard or sighted within the same habitat were recorded to the nearest metre along the transects. The number of days that tracks had accumulated (days since last snowfall to the nearest 0.5 day) was also recorded.

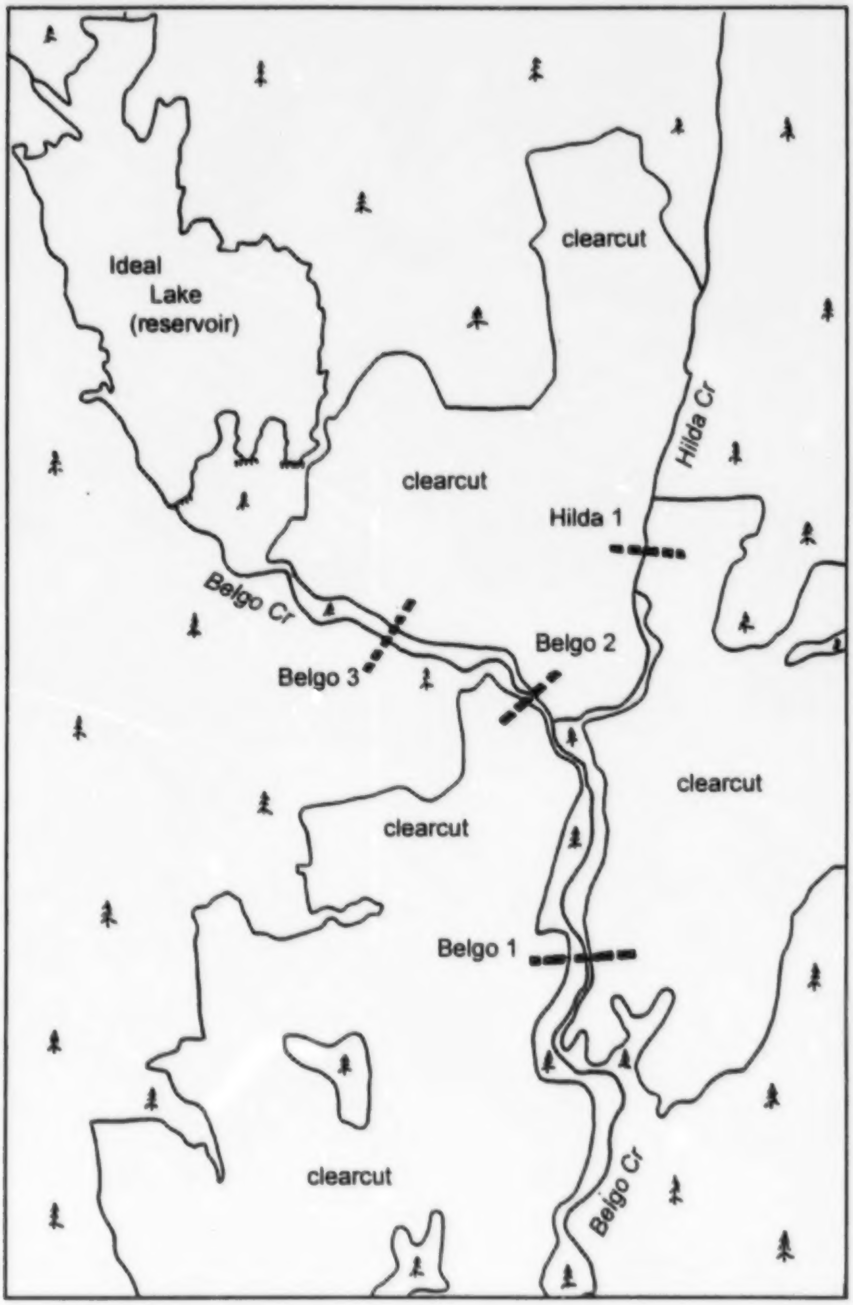
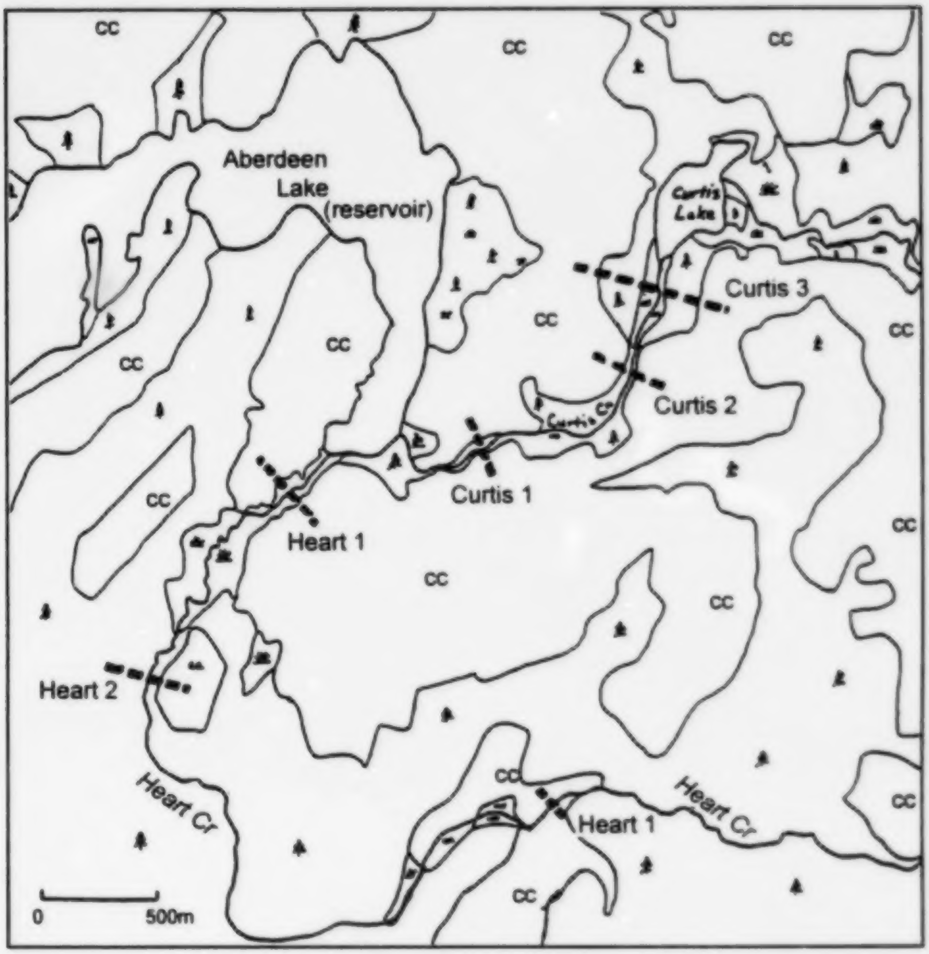

Ideal
Lake
(reservoir)
clearcut
clearcut
Hilda Cr
Hilda 1
Belgo Cr
Belgo 2
Belgo 3
clearcut
clearcut
Belgo 1
clearcut
Belgo Cr

Figure 13. Winter wildlife study sites along Belgo Creek and Hilda Creek near Ideal Lake, British Columbia.


cc
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Aberdeen
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Heart 1
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Figure 14. Winter wildlife study sites along Heart Creek and Curtis Creek near Aberdeen Lake, British Columbia.

Table 18. Habitat use (tracks/km-day) for common winter wildlife in the MSdm1.

Treatment of Riparian Management Zone	N	Habitat use (tracks/km-day)							
		Marten		Red Squirrel		Snowshoe Hare		Weasel	
		Mean[1]	S.E.[2]	Mean	S.E.	Mean	S.E.	Mean	S.E.
Total reserve	3	2.4	1.6	11.7	6.4	10.0	3.5	5.7	3.6
Partial harvest (25-33% removal)	3	3.2	0.6	15.5	0.9	7.7	3.6	1.8	0.3
No-machine buffer (clearcut)	3	0.4	0.4	0		2.1	2.1	10.2	5.1
ANOVA statistic for the above habitats: $F_{2,6}$; P		2.06; 0.21		4.74; 0.06		1.68; 0.26		1.37; 0.32	
OTHER HABITATS									
Partial harvest (50-85% removal)	1	4.2		7.4		2.6		5.3	
Clearcut (<10 years)	9	0		0.1	0.1	0.2	0.4	4.0	1.8
Clearcut (14 years)	1	0		10.8		4.0		9.8	
Unharvested forest	2	1.3-3.2		3.3-3.5		12.3-14.2		2.0-3.5	
Riparian forest	1	2.4		15.1		6.4		3.2	
Riparian zone (marsh or creek surface)	2	0		0		0		0.4-4.7	

[1] Where N=2, the range is given.

[2] Where N<3, no standard error (S.E.) was calculated.

For data analysis, the number of tracks for each species were summed for each site and each habitat over the whole winter and then divided by the product of transect length and accumulated days since snowfall. The resulting statistic is expressed in tracks/km-day, and is commonly used for quantifying winter track counts (Gyug 1988, 1994; Thompson et al. 1989). This measure is used as an index of habitat use and suitability, even though it is also probably related to population density.

The number of birds observed were tallied by site, species, and habitat type. To avoid double counting birds on side-by-side transects, only birds within 50 m of a transect were counted. Too few birds were encountered to attempt statistical analysis, although the results are presented for examination here.

The extent of the data analysis was limited by the number of sites available for examination. Even though four track segments were conducted at each site from the creek out to the clearcut, these were not independent samples. The habitat-use index calculated from the tracking totals in a given habitat type at each site was counted as an independent observation (only the most indicative of differences in wildlife use between treatment types will be detectable with the experimental design that was used). Because there were only three replicates of any given habitat (except for upland clearcuts adjacent to RMZs), track counts were supplemented with observations on behaviour of species based on track patterns near the sampled transects.

6.4 Results

A total of 1225 tracks were counted on snow-tracking transects. Species encountered (number of tracks) were: Weasel (385), Red Squirrel (342), Snowshoe Hare (301), Marten (75), Coyote (61), mice or voles (40), deer (14), Lynx (5) and grouse (2). Mice or voles could not be identified to species by tracks. No analysis of mice or vole tracks were attempted, since they are mainly active under the snow and are best sampled by other means. The grouse tracks could have been from any of the three species found in the region: Spruce Grouse, Blue Grouse or Ruffed Grouse.

6.4.1 Common species

Red Squirrels showed high use of 25-33% removal, partially cut sites and reserve-treatment sites, but no use of NMB treatment sites (Table 18). Red Squirrels were occasionally found in the clearcuts adjacent to the RMZ, but only in sites where tall shrubs predominated. The 14-year-old clearcut at green-up showed use by Red Squirrels equivalent to the use in reserve and 25-33% removal, partially cut RMZ sites. Although clearcuts can be considered non-habitat for Red Squirrels, they will repopulate a clearcut at a very early stage in its regrowth.

Snowshoe Hares showed very low use of clearcuts in which there was very little hiding cover and above-snow forage. They were found using only one of the three NMB areas. The tracks in this NMB area (Heart

1) indicated that it was used primarily as a travel route, since no heavily used hare trails or back-and-forth wanderings of foraging hares were seen. Hare use was highest in the reserve and forested sites, with slightly lower use in the partially cut sites. The partially cut stands appeared more open and with much lower hiding and foraging cover. This lack of hiding and foraging cover may have reduced the appeal of these habitats to hares. Even though clearcuts are essentially non-habitat for hares, considerable use was made of the single stand at green-up, so they will repopulate a clearcut at a very early stage in its regeneration.

Marten showed no use of clearcuts (even at green-up) on any of the transects, even though they were commonly found in the forest, reserve, and partially cut treatment sites. Marten appeared to use both the reserve and partially cut sites for hunting, as evidenced by the back-and-forth pattern of tracks and the tracks that led under downed logs and overturned stumps near the transects. However, landscape-level effects may be influencing RMZ use by Marten as well as stand-level effects. For example, the reserve site at Belgo 1 was not used by Marten in the tracked period, although Belgo 2 was used. These sites are both along the same 2.1-km forested RMZ corridor joining large forested areas. Belgo 1 is 500 m and Belgo 2 is 200 m from the nearest large forest. Perhaps this extra distance from continuous forest limited the use that Marten made of this forest corridor.

Additional landscape-level effects on Marten were found in the NMB sites. Only two Marten tracks were ever found in a NMB site (Heart 1). Both tracks were of the same Marten as it travelled parallel to the creek within the NMB treatment area. The Marten was using this area as a travel corridor of approximately 600 m between the forests to the south (near Heart 2) and the north (near Curtis 1). Marten prefer to stay in forests and would probably prefer forested corridors for travel between areas, but the nearest such corridor connecting these areas was 3.5 km to the east. Since it is unusual for Marten to cross openings of more than 250 m (Snyder and Bissonette 1987), this NMB RMZ was used only because no other options were available to get from one area of suitable habitat to another.

For the forest species common only in reserve or partially cut RMZs (Red Squirrels, Snowshoe Hares and Marten), we examined the response of habitat use (tracks per day) against the width of the RMZ as measured from stream bank or marsh edge to the edge of the clearcut bounding the RMZ (Figure 15). Only reserve RMZ sites were used for this comparison for Snowshoe Hares, as they were slightly less common in partially cut RMZ sites. The width of the RMZ was linearly correlated with habitat use for each species.

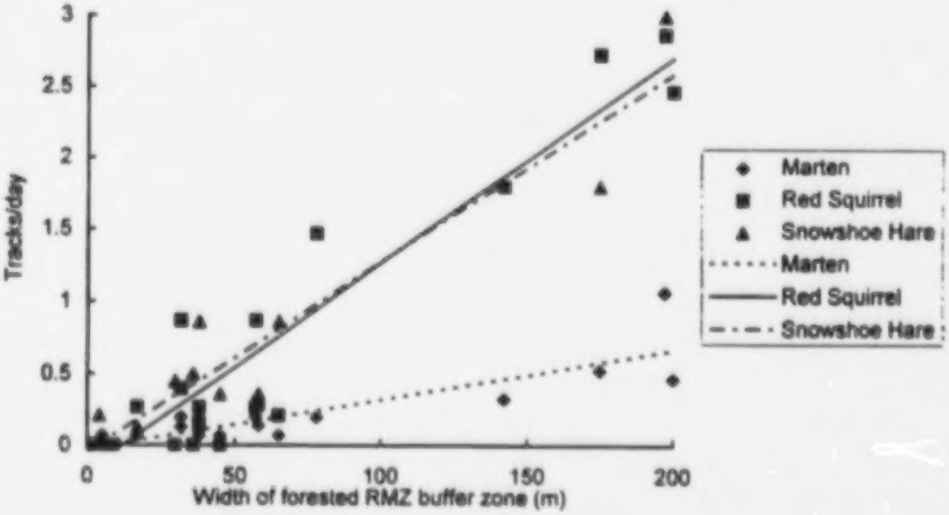


Figure 15. Red Squirrel, Snowshoe Hare, and Marten habitat use (tracks per day) against width of select-cut and reserve RMZ buffer as measured from creek or marsh edge to clearcut.

The linear regressions for each species were: Marten, $y = 0.035x - 0.027$, $r^2=0.78$, $F_{1,24}=29.21$, $P=0.000$; Red Squirrel, $y = 0.014x - 0.173$, $r^2=0.90$, $F_{1,24}=211.44$, $P=0.000$; Snowshoe Hare, $y = 0.013x - .044$, $r^2=0.92$, $F_{1,17}=197.66$, $P=0.000$.

The *y*-intercept was significantly different from the origin (0,0) only for the Red Squirrel ($t = -2.24$, $P=0.035$). Therefore, within the variability of the habitat-use index, only Red Squirrels can be distinguished as needing a minimum corridor width to sustain habitat suitability. The width for Red Squirrels was 12.0 m from the stream, which would be equivalent to a forested corridor of 25 m total width when counting both sides of a stream.

The weasel tracks encountered were probably all made by Short-tailed Weasels, since they appeared to be the only common weasels in the area. Weasel use showed no consistent statistical differences among the three RMZ harvest treatments when they were considered as whole units (Table 18). The highest weasel use was found in the NMB areas, even though one (Heart 3) showed no weasel use. When the weasel track data was re-examined based on distance from stream bank or marsh edge in 10-m increments (Figure 16), there was a definite peak of activity immediately adjacent to the clearcut riparian areas. However, weasel tracks were also found throughout the clearcuts and forests.

6.4.2 Birds

A total of 30 birds were seen or heard during the track transects (Table 19). No birds were seen in young clearcuts, except one Red Crossbill at Belgo 2, where some non-merchantable lodgepole pine had been left on a steep, rocky slope. Mountain Chickadees (six in three groups) and a Black-capped Chickadee were seen in the 14-year-old clearcut. Common Ravens were seen in the areas on most visits, but none were counted in the sampled habitats during the actual transect counts.

No birds were detected in the NMB areas or surrounding clearcuts. Only one Mountain Chickadee was counted within the 75% removal partially cut area. However, Mountain Chickadees, Golden-crowned Kinglets, and a woodpecker were seen or heard in the low-level removal partially cut areas, and woodpeckers, a Gray Jay, and Red Crossbill were counted in the total reserve areas.

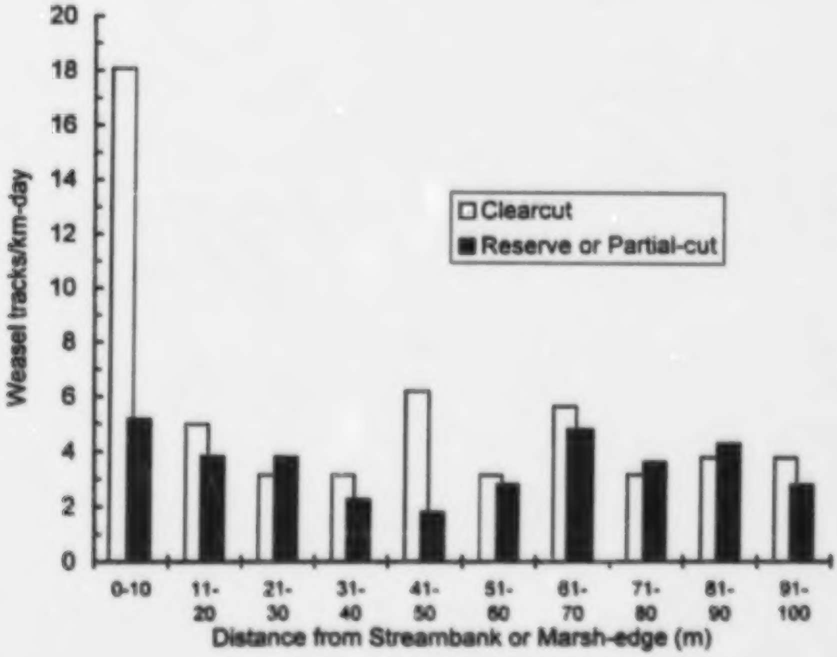


Figure 16. **Weasel habitat use (tracks/km-day) compared to distance from creek or marsh edges.**

Table 19. Number of birds detected on winter transects in different habitats adjacent to streams in the MSdm1.

					Number of birds detected of each species[1]						
Treatment of RMZ	Sampling area (ha)	No. of visits	Total area sampled (ha)	Mean bird detect's /ha	TTWO	wdpk	GCKI	GRJA	MOCH	BCCH	RECR
Total reserve	3.22	5	16.10	0.31	2	1	0	1	0	0	1
Partial-harvest											
25-33% removal	7.63	5	38.15	0.34	0	1	4	0	8	0	0
50-85% removal	1.29	5	6.45	0.16	0	0	0	0	1	0	0
No-machine buffer	1.56	5	7.80	0	0	0	0	0	0	0	0
	0.25	3	0.75								
OTHER HABITATS											
Clearcut (<10 years)	27.54	5	137.70	0.006	0	0	0	0	0	0	1
	5.81	3	17.43								
Clearcut (14 years)	4.00	5	20.00	0.35	0	0	0	0	6	1	0
Unharvested forest	3.26	5	16.30	0.08	2	0	0	0	0	0	0
	2.89	3	8.67								
Riparian forest	0.84	5	4.20	0.24	1	0	0	0	0	0	0
Riparian zone (marsh, creek surface)	2.49	5	12.45	0	0	0	0	0	0	0	0

[1] TTWO = Three-toed Woodpecker, wdpk = unidentified woodpecker, GCKI = Golden-crowned Kinglet, GRJA = Gray Jay, MOCH = Mountain Chickadee, BCCH = Black-capped Chickadee, RECR = Red Crossbill.

6.4.3 Uncommon species

Coyotes showed no clear response to the different harvest treatments of the RMZ. Travel routes tended to be parallel to the stream, so that coyote trails usually crossed both of the transects at any site. They seemed to prefer easy travel routes wherever they occurred. These included snow-covered roads, frozen creek surfaces with shallow snow, flat microsites on gully bottoms, on terraces above the creek gullies, or on flat open marshes when the snow crusted. The treatment (clearcut, partially cut, or reserve) did not appear to affect where these travel routes were. Topography and ease of travel (particularly the lack of impeding shrubs) seemed to play a larger role.

Deer tracks (White-tailed and/or Mule deer) were only found on the first visits in early December. Deer tracks were found in the clearcuts, the reserve, and the partially cut sites. Since the area is not deer winter range, these treatments probably have little effect on winter deer use. Moose tracks were found in the vicinity of Curtis Creek throughout the winter, but never on the transects. Moose tracks were seen in the Belgo Creek area until mid-January, but no tracks were found on the transects.

The five Lynx tracks were all from one animal that used both sides of the Curtis 3 partially cut site, probably while hunting for Snowshoe Hares. This low sample size was too low to permit making any conclusions about treatment effects on Lynx. Since the main prey of Lynx is Snowshoe Hares, then any treatment that benefits Snowshoe Hares has the potential to benefit Lynx.

6.5 Discussion and Management Implications

6.5.1 Riparian to upland comparisons

None of the common species for which we gathered data have a distinct reliance on riparian areas, i.e., they were all also found in upland areas. It remains to be shown whether upland forested corridors would be just as effective as any riparian forested corridors for these species. For Red Squirrels and Snowshoe Hares, upland corridors would probably be just as effective. However, Marten may have a reliance on riparian forests in winter.

The critical habitat requirement for Marten in winter are downed logs or small live spruce or fir trees to gain entrance to their prey under the snow (Corn and Raphael 1992). If downed logs and small trees are more common along riparian corridors then along similar upland corridors, then these riparian corridors will be of more value to Marten. Upland forests tended to be almost pure lodgepole pine stands with very little understorey (see section 2), while riparian forests had more intermediate canopy and understorey trees. Buskirk et al. (1989) also found that Marten winter resting sites tended to be closer to streams and lakes than a random distribution, that the resting sites in spruce-fir forest were more often associated with CWD than in lodgepole pine stands, that mean CWD size was larger in spruce-fir forests and often consisted of large upturned rootballs found almost exclusively in spruce-fir stands, and that Marten rested longer at sites associated with CWD than other types of resting sites. These findings all indicate that preferred Marten winter resting habitat will be spruce-fir forest. In the MSdm1 of the Okanagan Highlands, these sites are almost entirely associated with riparian forests (see section 2).

Extremely high weasel use of clearcut riparian areas coincided with our finding of high numbers of Meadow Voles at these sites (see section 2). Weasels will concentrate activity where hunting is most profitable, i.e., where prey is accessible, easy to catch (usually voles), and abundant (King 1989). However, Meadow Voles were not abundant at all riparian clearcut sites, so that this may have contributed to some of the variability in weasel track densities found here.

6.5.2 Effects of partial timber harvest

For winter mammals that are common above the snow, total reserve or 25-33% removal, partial cutting of the RMZ is preferable for Marten and Red Squirrels; total reserve is preferable, although not essential, for Snowshoe Hares; the difference between total reserve and partial cutting did not appear to influence weasels or Coyotes.

Marten used partially cut areas in RMZs at the same levels that they used total reserves. In other studies in upland areas, Marten have used partially cut sites at similar levels to forested sites in Idaho and Maine (Koehler et al. 1975; Soutiere 1979), and this probably extends as well to southern B.C. (L. Gyug, pers. obs., Kettle Valley, February 1993). However, although Marten hunted in both the partially cut RMZs and the total reserves, it is not known if there are suitable winter denning and resting sites in the partially cut RMZs.

6.5.3 Effects of forest corridor width and forest fragmentation

For Marten and for Snowshoe Hares, any forested corridor appeared to be used in direct proportion to its width, although there was some use in proportion to the corridor length, with areas further along a forested corridor less used than areas closer to the main block of forest. A more precise delineation of the use of forested corridors of different lengths would need a study of wider scope designed to determine landscape-level effects rather than just stand-level effects.

Marten only used clearcut RMZs as travel corridors within areas where the forest was highly fragmented (see Figure 14), and they probably spent considerable time and energy travelling from one suitable foraging habitat to another. Marten use of riparian corridors within large clearcuts was shown to be proportional to the amount of cover provided by logging debris piles, and reserved trees or understorey along streams or gullies (Gyug 1994). Additionally, one of the total reserve sites along Belgo Creek showed no use, even though it appeared to be suitable habitat. Fragmented forest may therefore be sub-optimal habitat because of its fragmentation. Without detailed habitat-use information gained from trailing or radio-telemetry, the true usefulness of partially cut sites and estimation of optimal habitat cannot be adequately addressed.

For both Red Squirrels and Snowshoe Hares, clearcuts at the green-up stage (>3-m height of conifers) regain a large measure of habitat suitability over more recent clearcuts. These clearcuts appeared usually to be repopulated within 15 years by these two species (assuming nucleus populations existed nearby to provide immigrants). The present patterns of timber harvest often rely on RMZs as these nuclei for repopulation, so that reserve or partial cuts along streams and wetlands would be the preferable options in these cases.

6.5.4 Birds

Birds were uncommon in the Montane Spruce Zone in mid-winter. No nuthatches or Brown Creepers, and no flocks of finches were encountered. For those species that did use the area, no birds were encountered in the clearcut riparian zones. No woodpeckers were encountered in the heavily partially cut areas probably because all feeding trees (snags and insect-attacked trees) had been recently felled. However, woodpeckers will feed in areas immediately adjacent to these as long as the standing stems that contain insects are not felled. Given the present high infestation levels of

Mountain Pine Beetle, there is high potential for insect attack within the partially cut areas, and Three-toed Woodpecker feeding trees may quickly become more common there. However, nesting sites will still be lacking for cavity-nesting birds in these partially cut areas. For the highest diversity and numbers of overwintering birds, a low-level partial cut (25-33% removal) or a total reserve is preferable along an RMZ.

6.5.5 Uncommon species

For winter-active mammals that are uncommon and for which there is presently little information, the most appropriate harvest treatment of the RMZ can only be estimated. Lynx will likely be best served by any treatment that preserves high numbers of Snowshoe Hares, their principal prey. However, because Snowshoe Hares quickly repopulate clearcuts, management of RMZ for Lynx may not be essential.

No data were collected on flying squirrels, and their abundance is unknown in the area. Since flying squirrels appear to depend on tall trees and on cavities in trees as nests, a reserve treatment may be preferred to a partial-cut or NMB treatment within the RMZ. Witt (1992) has indicated that flying squirrels are scarce or absent in second-growth stands (50-65 years old), but more abundant in old-growth stands in western Oregon. Flying squirrels may be more common in second-growth stands in the montane zone in the Okanagan (D. Ransome, UBC, pers. comm.) but their habitat preferences and response to partial cutting have yet to be fully determined.

None of the species considered so far have an absolute dependence on riparian areas. Each of these species is also common in upland forest areas. For instance, the response of habitat use to width of RMZ buffer would certainly not have been linear for riparian-dependent species such as Mink, otter, Beaver, and Muskrat, as it was for Red Squirrels, Marten, and Snowshoe Hare. However, none of these riparian species were found on the track transects. For these species, the sampling design and analysis would best be set up with tracking transects running parallel at different distances from the riparian area, and study sites chosen in areas that have (or had) countable populations of these species.

Moose will use a variety of habitats in winter wherever forage is abundant and accessible — very often in riparian zones. This does not preclude NMB treatment of the RMZ as long as some reserve, or perhaps also partially cut, RMZ is nearby for security, for thermal cover, and to provide areas of moderate snow depths at critical periods. Since Moose are wide-ranging, the number of stands examined must be quite high to ensure adequate data, and data collection should be aimed specifically at Moose winter range areas.

7 INTERNAL VALIDATION AND FPC RIPARIAN GUIDELINE APPLICATION IN TWO DRAINAGES

7.1 Introduction

This part of the study was undertaken to verify the predicted levels of trees and coarse woody debris, breeding birds, and amphibians found in the main study (sections 2 to 6) in riparian ecosystems in the Okanagan Highlands of the southern interior of British Columbia. Two small (900-1300 ha) drainages were selected and sampled extensively for breeding birds, trees, and CWD, and these levels were compared to the levels predicted from the main study. If there were no significant differences between the levels found in the main study and in the levels found within the two drainages sampled here, then there can be a large degree of confidence in the results, and consequently a large degree of confidence in the patterns suggested by the results. Fairly extensive sampling in these two drainages also enabled examination of the responses of bird species and tree species to a wider variety of riparian situations than were investigated in the main study, where attempts were made to minimize variability between sites to minimize "noise" in the experimental design.

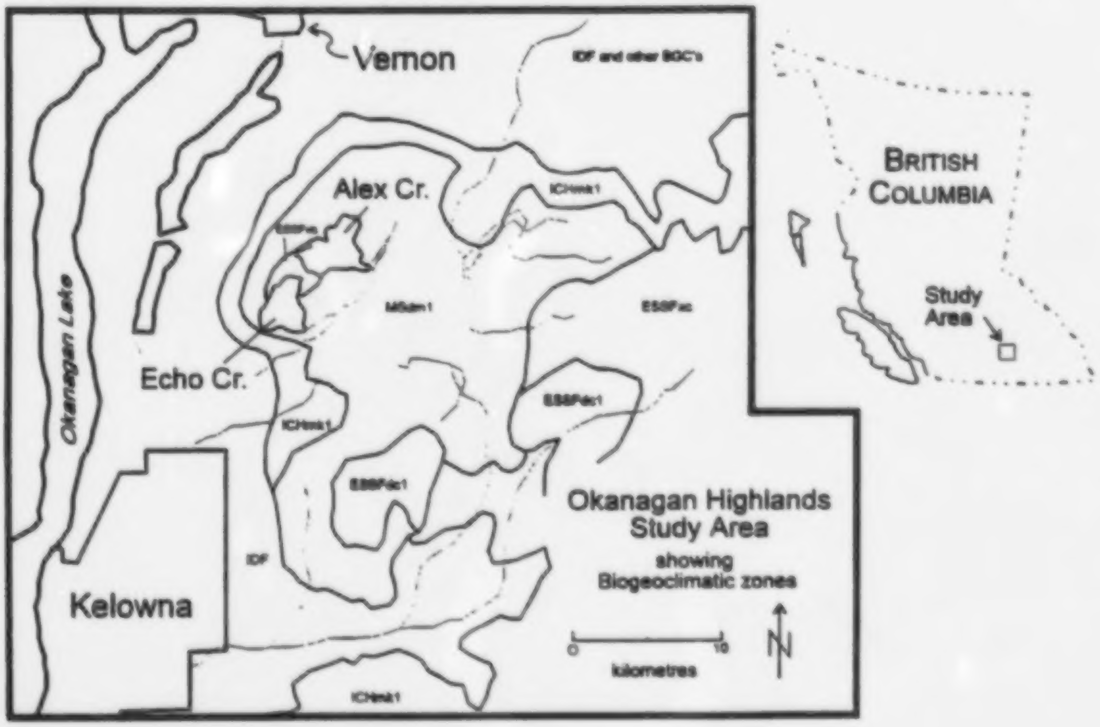
The specific objectives of this part of the study were to determine the levels of the following parameters in the Echo Creek and Alex Creek drainages of the Okanagan Highlands, and compare them statistically to the levels found in the main study:

- The forest stand species composition and abundance, and the effects of stand age on stand composition in unharvested upland and riparian areas.
- The density and dispersion of trees that survived the previous major fires in riparian and upland areas.
- The levels of CWD in relation to stand age and riparian or upland position.
- The density and woodpecker use of wildlife trees in relation to species, stand age, and riparian or upland position.
- The densities of breeding-bird detections in relation to forest or clearcut habitat, and riparian and upland situation.

Secondary objectives were to determine how the riparian management guidelines as outlined in the FPC riparian guidelines would be applied to the two selected drainages, and to assess the guidelines with respect to the findings of this study.

7.2 Study Area

The study area for this project was in the Okanagan Highlands east of Kelowna, B.C. (Figure 17). Two small second-order drainages were selected for intensive study: Echo Creek (917 ha, Figure 18) and Alex Creek (1263 ha, Figure 19). These drainages are in the Vernon Forest District of the Okanagan Timber Supply Area of the Kamloops Forest Region of the B.C. Ministry of Forests. The drainages were selected because first-pass clearcutting has been completed, so that both clearcut and unharvested forest habitat were available for wildlife assessments, and both contained only the two biogeoclimatic zones that were the focus of the main study: the Okanagan Dry Mild variant of the Montane Spruce Zone (MSdm1) at elevations from 1350 to 1500 m in the drainages, and the Very Dry Cold subzone of the Engelmann Spruce–Subalpine Fir Zone (ESSFxc) at elevations from 1500 to 1620 m (Lloyd et al. 1990). Sixty-nine percent of the Echo Creek drainage and 61% of the Alex Creek drainage were in the MSdm1. Non-forest land, either lakes, marsh, or rock bluffs covered 3.9% of the Echo Creek drainage and 2.9% of the Alex Creek drainage. Twenty-two percent of the Echo Creek drainage and 34% of the Alex Creek drainage had been clearcut at least eight years before this study. The Echo Creek clearcuts were small (10-20 ha) and relatively evenly dispersed. The Alex Creek clearcuts were originally small and dispersed as in Echo Creek, but the opening sizes have expanded because of clearcutting to control Mountain Pine Beetle. The two largest openings are now 117 and 227 ha (considering only the parts of the clearcuts within the drainage). No clearcutting has taken place in the last eight years.


Vernon
IDF and other BGC's
Alex Cr.
ICHmk1
ESSFxc
MSdm1
ESSFxc
Okanagan Lake
Echo Cr.
ESSFdc1
ICHmk1
ESSFdc1
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ICHmk1
Okanagan Highlands
Study Area
showing
Biogeoclimatic zones
0
10
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COLUMBIA
Study
Area

Figure 17. Two selected drainages in the Okanagan Highlands of southern British Columbia.

7.3 Methods

7.3.1 Sampling methodology

Breeding birds were sampled along transects chosen along the lake boundaries, along most of the streams, and adjacent to most of the wetlands within the study drainages. Since riparian habitats were the focus of this study, sampling of all major riparian sites in the drainages was the first priority. Upland transects of about half the total length of the riparian transects were also chosen. Total areas sampled for birds were: open water 11.3 ha, non-forested wetlands 19.2 ha, riparian clearcuts <20 years old 34.1 ha, upland clearcuts <20 years old 62.8 ha, upland MSdm1 clearcuts 20-25 years old 18.0 ha, unharvested riparian forests 150.9 ha, and unharvested upland forests 73.3 ha. Sampling in each drainage, and in the MSdm1 and ESSFxc of each drainage, were proportional to the areas occupied by each. Each transect was sampled once by an observer walking slowly (1-2 km/hr) between 0430 and 1100 hours from 25 to 28 May 1995. The species, activity, and habitat (forest, clearcut, marsh, swamp, and open water) of all birds detected within 50 m of the transects were recorded. Any amphibians encountered were also noted.

Other parameters (stream physical measurements, stand age, trees, and CDW) were estimated at a total of 50 points established about every 400 m on the forested portions of the breeding-bird transects. Sampling took place in September 1995. At each point, a 0.2-ha (20 × 100 m) main plot was established. For streamside plots, the stream formed the main-axis centreline of this plot. Along wetlands and lakes, the plot extended 20 m upslope from the non-forest/forest boundary. Within the main plot, a subplot 5-m wide was established along one side of a stream or at the non-forest/forest boundary for wetlands and lakes. The length of the subplot was dictated by the number of live trees counted. After 25 live trees were tallied, the plot was ended and the length recorded. A 45-m CWD transect was established within the main plot about 2.5 m from the stream or non-forest/forest boundary.

The physical characteristics recorded for each site included stream width and "active floodplain" width at six points separated by approximately the bank-to-bank width. "Active floodplain" was defined in the FPC riparian guidelines as the approximate equivalent of annual floodplain, or active channel. Valley bottom width was measured.

At each point, the time since the last major disturbance event was determined by coring trees from the dominant cohort of canopy trees, and was termed "stand age." At all sites, fire could be identified as the previous major disturbance event that had given rise to the dominant cohort of canopy trees. Possible veteran trees that may have survived the last major disturbance were identified and also cored to determine if they were veterans. At most sites, a diameter limit could be found that differentiated veterans from trees that had grown since the last major disturbance. Questionable veteran trees were cored to determine age accurately. A census for veteran trees in the 0.2-ha plot was then taken. Twenty-five living trees >10 cm dbh were tallied in the subplot. Data collected for each tree were: species, canopy position or strata (veteran tree, canopy tree, or sub-canopy tree), and dbh to 10-cm classes (10-19.9 cm, 20-29.9 cm, etc.).

Wildlife trees (dead standing trees) ≥30 cm dbh were tallied in the 0.2-ha plots. All wildlife trees >10 cm dbh were tallied in the subplots. Only wildlife trees of classes 3 to 7 (Province of British Columbia 1995b) were counted. Dead trees <0.5-m height were not counted, nor were dead trees <1.0-m height that had no part of the outer wood surface (the surface immediately under the bark) remaining intact. These were considered to be class 8 wildlife trees and were not tallied. Data collected for each tree were: species, dbh to 10-cm classes (10-19.9 cm, 20-29.9 cm, etc.), height, wildlife tree class, probable cause of tree death if readily identifiable, and signs of animal use. Species of animals using the tree were noted where possible, and time since use was estimated as recent or older. Recent use was characterized by freshly exposed wood, and probably indicated use within the previous 8 to 12 months.

Downed logs, or CWD, ≥7.5 cm diameter were tallied on a 45-m intercept line parallel to the stream, wetland, or lake and 5 m distant. Diameter and decay class (1-4 of Maser et al. 1979) of each piece intercepting the line were noted. The source of each piece of CWD, i.e., developed from trees grown in the pre-fire stand or from trees grown after the last major fire, was estimated.

7.3.2 Data analyses

The purpose of all the analyses was to compare various parameters found in the Echo/Alex drainages to the results given in other parts of this report. Most comparisons (particularly those of birds) were made by examination of the data: if a given parameter fell

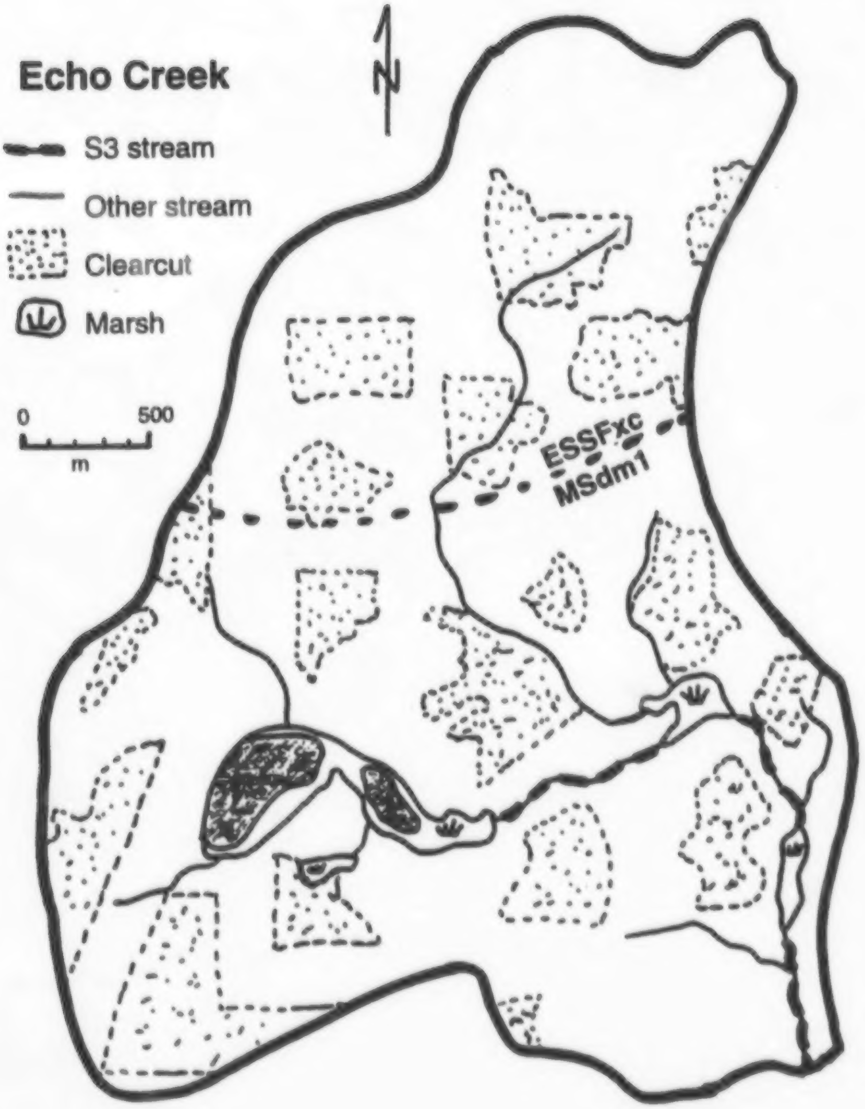
Echo Creek
S3 stream
Other stream
Clearcut
Marsh
0
500
m
N
ESSFxc
MSdm1

Figure 18. Echo Creek drainage, showing biogeoclimatic zones, streams, marshes, and clearcuts.

outside the 95% prediction limit (mean ± $t_{(\alpha/2)}$ × Std. Dev.) developed elsewhere in the report, then it was judged to be significantly different. Parameters for which there was a large sample size (particularly tree density and CWD data) could be compared between the other studies and the Echo/Alex drainage sub-study using *t*-tests (and adjusting the test for non-homogeneous variances when necessary).

Since sites were not randomly chosen, they could not be used for estimating a stand age distribution to compare to the other studies. Even though the areas were fairly small and ages were sampled at 50 sites, "fire-history" mapping could not be completed using the forest cover mapping because the estimated forest cover age classes were found to be very inaccurate. This accuracy was compared to the accuracy found in the other studies using *t*-tests.

Live tree density estimates (trees/ha) were calculated for each site for each species and strata, and compared to the other studies using *t*-tests. Densities of veteran trees were based on the 0.2-ha plots, and all other densities were based on the smaller subplot at each site. To examine the total tree structure of the stand, the ratio of canopy trees to sub-canopy trees was calculated and examined for streamside to upland and BGC differences using ANOVA. Since no sub-canopy trees were found at some sites (particularly those dominated by lodgepole pine), these sites were arbitrarily assigned a canopy to sub-canopy ratio of 25 (since the samples were made of only 25 trees). Mean tree size (dbh) of the canopy layer was also calculated for each site, and for each species at each site, and compared between upland and streamside sites. However, because size was expected to vary with stand age, analysis of covariance (ANCOVA) methods were used for this comparison.

Standing dead tree (wildlife tree) density estimates (trees/ha) were calculated for each site for the 10-29.9 cm and ≥30 cm dbh classes. The density estimate for small wildlife trees (10-29.9 cm) was based on the subplots, and, for large wildlife trees (≥30 cm), they were based on the 0.2-ha plots. Density of wildlife trees showing new woodpecker or cavity-nester use was calculated for the small and large trees separately, and percentage of each class showing use was also calculated. Total use was also tallied for each species and each size class of trees, and the rate of tree use was compared using χ^2 contingency tables.

Wildlife tree density was found within the main study to be a complex function of the numbers and sizes of trees killed in the stand-initiating fires, as well as the numbers of trees developed from within the post-fire cohort — both of which are a function of stand age. Wildlife trees in the Echo/Alex study were categorized as developing from within the present cohort, or being remnants of the previous stand. The number of wildlife trees remaining from pre-fire stands was presented, but not analyzed because they were found too infrequently. The development of wildlife trees from within the post-fire cohort was analyzed using ANCOVA with $\log_{10}$(stand age) as the covariate, and with riparian/upland position and study (i.e., the other studies or Echo/Alex drainage study) as the fixed factors to determine if there were significant differences between them.

Volumes of downed CWD/ha in m^3/ha were estimated using the formula from Harmon et al. (1986): Volume = $\pi^2\sum d^2/8L$, where d=log diameter and L=transect length. Volumes of CWD were found, in the main study, to vary significantly with $\log_{10}$(stand age), and, as for wildlife trees, were a complex function of time-since-fire, the falling and decay rates of trees killed in the fire, and the volumes developed from within the post-fire cohort. Not enough young stands were sampled in the Echo/Alex drainages to statistically compare the decay of CWD volumes after a stand-destroying fire. CWD volume developed from within the post-fire cohort was analyzed for streamside to upland differences using ANCOVA, with $\log_{10}$(stand age) as the covariate, and then were analyzed for between-study differences, again using ANCOVA.

7.4 Internal Validation

7.4.1 Stream physical characteristics

In the Echo and Alex drainages, widths of the first-order streams were not significantly smaller than second-order streams (0.78±0.12 m, N=16 vs. 1.33±0.36 m, N=6; $F_{1,20}$=3.54, P=0.07). The range of first-order stream widths was from 0.1 to 1.82 m and the range of second-order streams was from 0.5 to 2.9 m. Bank-full stream width was, on average, 51% of the active floodplain width as defined in the FPC riparian guidelines. Active floodplain widths of first-order streams were significantly narrower than second-order streams (1.52±0.23 m, N=16 vs. 2.65±0.43 m, N=6; $F_{1,20}$=6.22, P=0.02). Valley-bottom width was not significantly correlated with either stream width (r^2=0.02) or active floodplain width (r^2=0.00). As in the main study, the sizes of gullies and valleys seemed to bear little relationship to the size of the stream within them because most of the valleys appeared to be remnants of glacial meltwater channels. The extent of riparian forest cannot be predicted based on stream size. The only sure way to determine the extent of riparian forest is a site visit in

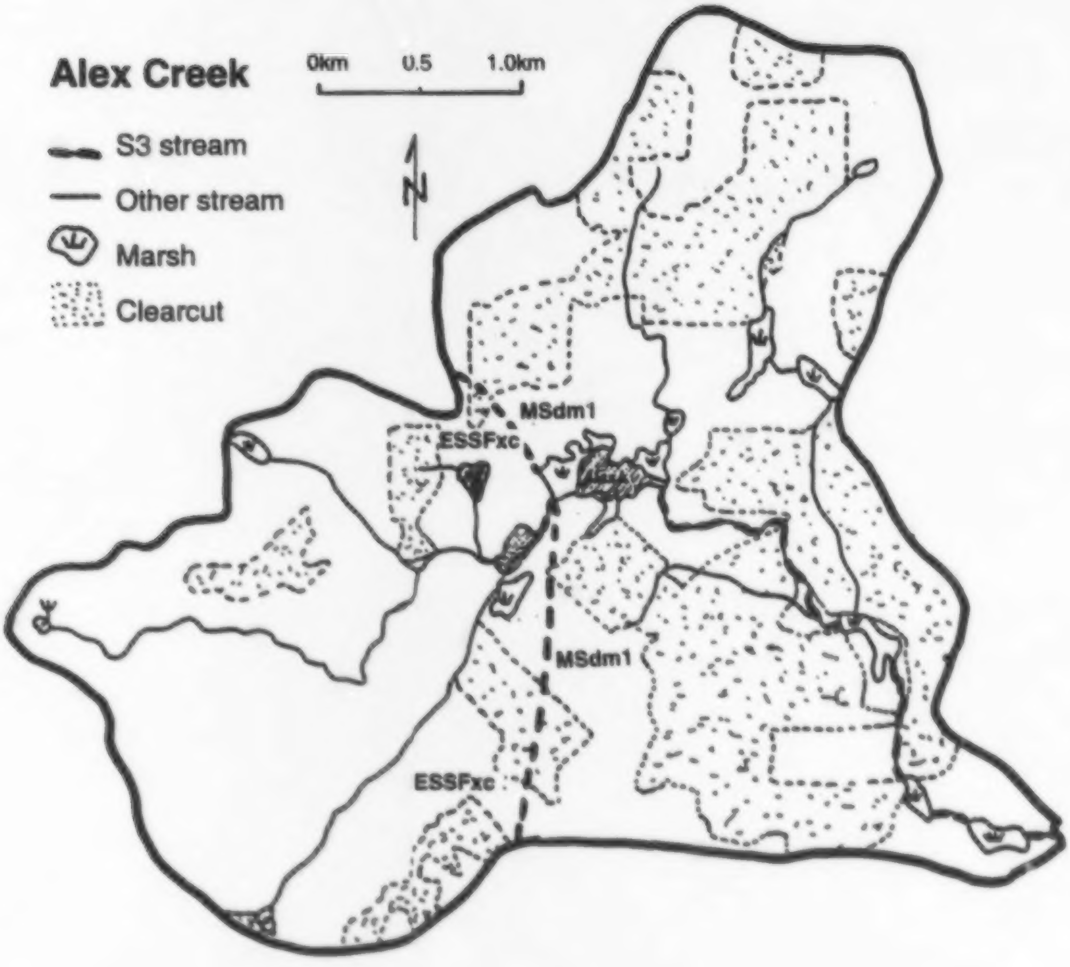
Alex Creek
0km
0.5
1.0km
S3 stream
Other stream
Marsh
Clearcut
N
MSdm1
ESSFxc
MSdm1
ESSFxc

Figure 19. Alex Creek drainage, showing biogeoclimatic zones, streams, marshes, and clearcuts.

the snow-free season (although aerial photo interpretation can also be useful).

7.4.2 Stand ages

Only one small 4.5-ha patch of old-growth forest (>250 years) was sampled for trees in the Echo/Alex drainages. This patch, on a wet seepage area, appeared to have survived a major fire 88 years ago that gave rise to the stands around this patch. Other than this one patch in Echo Creek drainage, and a few other very small patches in the Alex Creek drainage that were not visited (and therefore not precisely aged), there were no old-growth stands in the area. A series of fires occurring between 1825 and 1928 (based on lodgepole pine ages) appeared to have burned over the entire drainages. Between 1825 and 1928, at least 11 major fires gave rise to the present stands in the Alex Creek drainage. Between 1840 and 1896, at least 10 major fires gave rise to the present stands in the Echo Creek drainage. Aerial photo interpretation only reliably distinguished fire boundaries for fires that occurred <100 years ago (since 1895). Because a large portion of the stands appeared to be older than 100 years and could not be reliably separated without very intensive field sampling, no further estimations of stand age distribution or time-since-fire distributions were attempted.

7.4.3 Forest cover map errors in stand age estimation

The stand age classes of the forest cover mapping were found to be quite accurate in section 2 (57% — 26 of 46 sites mapped correctly), but were very inaccurate in the Echo/Alex drainages (20% — 10 of 50 sites mapped correctly). For those sites mapped in the main study as age class 8 (141-250 years), four of the 10 were correct, and six of the 10 were actually older (were >250-year-old stands). The bias in the mapping age for class 8 sites in the Echo/Alex drainages was exactly the opposite — 14 of the 16 sites were actually younger than 137 years (allowing four years as a margin of error for tree-coring ages) — and was significantly different from the main study (t=5.15; P=0.00). Only for two of the 14 sites did there appear to be an actual interpretation error (a large and obvious age class 3 stand was inexplicably divided into an age class 3 and age class 8 stand during forest cover mapping). The rest of the errors were simply nuances of interpretation based on stand height, composition, and density, which more accurately reflect topography and growing site conditions than stand age. Apparently, when large tracts of lodgepole pine forest reach 115-140 years in age, estimates of stand age based on height, composition, and density will not be very reliable. Similarly (from the main study), when large tracts reach ages >250 years, and are intermixed with younger stands, stand age estimation will not be reliable. Because the actual error in a mapped age class 8 stand can be significantly biased either upward or downward, age class 8 mapping from forest cover maps may be unreliable.

For stands mapped as younger than 140 years, the error in mapping was found to be the same in the main study and in the Echo/Alex drainages (t=1.19; P=0.24). The error was unbiased, i.e., there were as many sites misclassified older as younger, so that the map ages may be used for analyses without fear of introducing directional bias. In section 2, 94% of the sites were either correctly classified, or within one 20-year age class of being correct, compared to 79% in the Echo/Alex drainages.

7.4.4 Live trees

Only three common species of trees (lodgepole pine, spruce, and subalpine fir) made up 95.5% of the 1250 trees counted in the subplots. Both Engelmann spruce and Engelmann-white hybrid spruce occur in the area, and no attempts were made to separate out spruce trees to species. Other species encountered uncommonly were (in order of decreasing abundance): alder, Scouler's willow, Douglas-fir, and black cottonwood.

As in the main study (section 2), the canopy layer of young riparian stands (<170 years old) were dominated by spruce, and young upland stands by very dense lodgepole pine (Table 20). The sub-canopy layers were composed of both spruce and subalpine fir. There were no significant differences in tree species density in the young MSdm1 stands sampled in section 2 and in the Echo/Alex drainages. However, there were significant differences in the ESSFxc stands. Lodgepole pine was not abundant in the few ESSFxc stands (four) sampled in section 2, but was significantly more abundant in the Echo/Alex drainages, and much more similar in species densities to the MSdm1 (where spruce dominated in the riparian forests and lodgepole pine dominated in the upland). The single old-growth stand (300 years) sampled in Echo/Alex was similar in density and composition to the main study sites with lodgepole pine absent and overall tree densities lower than in the younger stands.

Table 20. Riparian to upland comparisons of live tree densities by biogeoclimatic zone, canopy position, and species, and of the ratio of canopy to sub-canopy trees. Figures that are significantly different from the main study are highlighted within boxes.

	MSdm1 young stands				ESSFxc young stands				ANOVA row-wise statistics[1]		Old-growth	
	Streamside		Upland		Streamside		Upland					
Parameter	Mean	S.E.	Mean	S.E.	Mean	S.E.	Mean	S.E.	$F_{3,42}$	P^2	Mean	S.E.
Live trees (>10 cm dbh) in Echo Creek and Alex Creek drainages												
(No. of sites)	(21)		(10)		(9)		(5)				(1)	
Canopy												
Spruce	346^{a}	68	31^{b}	21	240^{ab}	38	40^{b}	27	5.56	**0.00**	297	
Subalpine fir	93	36	14.5	9.9	173	77	100	45	1.59	0.21	297	
Lodgepole pine	473^{a}	94	977^{b}	111	452^{a}	85	998^{b}	265	5.45	**0.00**	0	
Other species	1.2	1.2	8.7	8.7	0	-	0	-	0.94	0.43	0	
Sub-canopy												
Spruce	367	107	87.6	23.5	204	40	107	65	1.82	0.16	186	
Subalpine fir	108	27	207	66	53	22	232	130	2.22	0.10	149	
Other species	64	32	10.4	7.0	125	63	0	-	1.63	0.20	0	
Canopy to sub-canopy tree ratio	3.4^{ab}	0.7	9.4^{bc}	3.2	2.7^{a}	0.4	10.9^{c}	5.8	3.41	**0.03**	1.78	
LIVE TREES (>10 CM DBH) IN MAIN STUDY (PART 2)												
(No. of sites)	(15)		(15)		(4)		(4)				(8)	
Canopy												
Spruce	360	62	77	32	323	67	314	124			169	25
Subalpine fir	92	32	16.9	5.7	83	14	264	88			198	44
Lodgepole pine	291	78	1159	197	44	31	102	43			7	4
Other species	6.6	3.5	8.5	7.7	0	-	0	-			0.8	0.8
Sub-canopy												
Spruce	206	57	114	30	230	66	191	84			0.77	20
Subalpine fir	104	28	63	21	173	69	209	46			253	20
Other species	43	22	21	16	1.6	1.6	1.6	0.6			2.4	1.6
Canopy to sub-canopy tree ratio	3.4	0.6	10.2	2.3	1.3	0.2	1.8	0.2			1.1	0.2
LIVE TREE COMPARISON STATISTICS BETWEEN THE TWO STUDIES[3]												
	t	P	t	P	t	P	t	P				
Canopy												
Spruce	0.11	0.91	1.04	0.31	0.71	0.49	1.99	0.14				
Subalpine fir	0.02	0.99	0.32	0.75	1.16	0.27	1.76	0.12				
Lodgepole pine	1.41	0.17	0.54	0.59	4.65	0.00	3.42	0.03				
Sub-canopy												
Spruce	1.33	0.19	0.82	0.42	0.34	0.74	0.80	0.45				
Subalpine fir	0.09	0.93	1.88	0.07	2.21	0.05	0.15	0.88				
Canopy to sub-canopy tree ratio	0.22	0.83	0.22	0.83	2.32	0.04	1.57	0.19				

[1] Row-wise means with like superscripts are not significantly different when row-wise P <0.05; Old-growth not included in comparison.

[2] Significant α denoted in **bold**.

[3] *t*-test between cell-by-cell study comparisons were adjusted when variances were significantly non-homogeneous.

The ratio of canopy to sub-canopy trees was similar in section 2 and the Echo/Alex drainage, except for ESSFxc sites (Table 20). The ratio was lowest in the old-growth stand, highest in young upland stands dominated by lodgepole pine, and intermediate in the riparian stands. The ratios of the Echo/Alex ESSFxc sites were more similar to the MSdm1 sites than to the ESSFxc sites sampled in section 2. This may simply be a factor of too few samples in the main study, and these samples may have been atypical. Only by more sampling in section 2 could this be resolved.

Within the cohort of canopy trees grown after a fire, mean tree size (dbh) was significantly correlated with $\log_{10}$(stand age) ($F_{1,47}$=12.46, P=0.001) and significantly affected by the riparian-upland position ($F_{1,47}$=22.44, P=0.000). Mean tree size in riparian stands was significantly larger than in upland stands with the streamside stands averaging 6.0 cm larger in mean dbh at stand ages of 120 years. The mean size of canopy trees in the riparian sites was described by the regression: Mean dbh = 28.46 × $\log_{10}$(stand age)

– 31.86; r^2=0.28. The mean size of canopy trees in the upland sites was described by the regression: Mean dbh = 17.04 × $\log_{10}$(stand age) – 14.14; r^2=0.18. The mean tree sizes also averaged 5.0 cm smaller in the upland MSdm1 sites in section 2 at stand ages between 100 and 120 years.

Several sites were left out of the above analyses. These were four sites along lakeshores or at the edges of marshlands where the valley bottom forest was <0.5 m wide. At these sites, the lakes or marshes butted almost directly up against steep banks two or more metres high. There was essentially no riparian forest at these sites, except for the odd tree growing directly on the lake or wetland margin. The sample plots, even at 5 m wide, were too wide in these situations to accurately reflect the width of the riparian forest, as these plots sampled more upland forest than riparian forest. At the four sites, lodgepole pine densities (1590/ha) were more than three times the usual for MSdm1 riparian sites, and spruce densities (140/ha) less than half that typical for riparian sites. However, even these spruce densities were more than three times higher than mean spruce densities in upland stands (see Table 20). These sites, therefore, are not typical of riparian forests, but are also not typical of upland forests. However, birds detected at these sites appeared to be very similar to those of riparian forests — whether because of proximity to lake or marsh, or because of the presence of the occasional mature spruce. Even in these extremely narrow riparian forests at the lake or wetland margins, some trees reached sizes of 40 cm dbh or greater, which was rarely ever achieved on upland stands in the area, and many reached 30 cm dbh or more, which was also uncommon in upland stands. These sites were therefore probably more typical of riparian stands than upland stands, even though 20-m wide plots contained elements of both.

7.4.5 Veteran trees

Veteran trees within the 0.2-ha sample plots were very uncommon in the Echo/Alex drainages, being found at only eight of the 50 plots sampled. Survival of veteran trees was not correlated with floodplain width (r^2=0.02). Mean overall density of veteran trees was 17.7/ha in the upland, and 3.1/ha in the riparian sites. While the mean appears much higher in the upland, the distribution was non-normal (just as in section 2), so that averages were not very meaningful, and the methods of calculation were not the same as in section 2 (i.e., upland-streamside plots were not paired, and the plots were smaller).

To make the data comparable between the two studies, riparian plots in the Echo/Alex drainages were defined as veteran tree patches if the upland stands <200 m upslope were older stands. This comparison could be made because the fairly intensive sampling in the Echo/Alex drainages (1 plot/43.6 ha) did allow fire boundaries and stand ages to be estimated, even for stands where no plots had been placed.

The distribution of veteran trees was very similar in the Echo/Alex drainages (Table 21) and in the main study (see Table 4). Similar lengths of stream had survived fires in patches >0.2 ha in both data samples (28% in section 2, 26% in the Echo/Alex drainages). Veteran trees surviving at densities of 10-20/ha were also very similar (11% in section 2, 13% in the Echo/Alex drainages). Veteran trees were found at fairly high densities (>40/ha) at 10% of the upland plots in section 2 and 13% in the Echo/Alex drainages.

Although the overall patterns of veteran tree distribution were the same in section 2 and the Echo/Alex drainages, there appeared to be some minor differences in veteran tree densities that were probably related to the difference in plot sizes. Larger plots have finer resolution than the 0.2-ha plots — densities below 5/ha will not be detectable, and veteran tree densities will always be estimated in multiples of 5/ha. Therefore, the two lowest-density classes (0 and 5/ha) should be grouped for the most reliable comparison. When these were grouped to allow for the difference in resolution, then the distributions were almost identical (Echo/Alex vs. section 2: 60% vs. 59% in riparian, and 87% vs. 86% in upland plots).

Table 21. **Veteran tree survival after fire in riparian and upland plots of the Echo and Alex drainages.**

Density and dispersion of veteran trees	% streamside length[1]	% upland area
N	3.5 km in 35 100-m plots	15 0.2-ha plots
In patches >0.2 ha	26	N/A[2]
Many scattered trees (40-225/ha)	3	13
Few scattered trees (10-20/ha)	11	Not found
Very few scattered trees (5/ha)	3	Not found
No veteran trees	57	87

[1] Streamside length based on area 10 m to each side of streambank.

[2] Not applicable because upland stand age used to define veteran trees.

7.4.6 Wildlife trees

As in section 2, in young stands there were no significant differences in the density or use of small (10-29.9-cm dbh) wildlife trees between riparian and upland sites in either the MSdm1 or ESSFxc (Table 22). The density of small wildlife trees in MSdm1 riparian stands was significantly higher in the Echo/Alex drainages than in section 2. Large wildlife trees (≥30 cm dbh) were significantly more abundant in young riparian MSdm1 sites than in young ESSFxc sites and young MSdm1 upland sites. As in section 2, the percentage of large wildlife trees recently used (only considering stands where they were available) was not significantly different between biogeoclimatic zones and topographic positions, indicating that birds used the large trees similarly wherever they were available.

As in section 2, woodpeckers in the Echo/Alex drainages used large wildlife trees significantly more frequently than small wildlife trees (χ^2=53.37, P=0.00, N=370 small trees and 77 large trees). For each species, recent use of large wildlife trees was higher than the use of small wildlife trees. For lodgepole pine, use of small trees was 6.8% compared to 29.4% for large trees (χ^2=17.49, P=0.00, N=236 small trees and 34 large trees); and for spruce 32.3% compared to 70.0% (χ^2=7.18, P=0.01, N=34 small trees and 20 large trees). No small Douglas-fir wildlife trees were counted, as in section 2, but each of three large Douglas-firs sampled had been recently used.

Among small wildlife trees, both spruce and subalpine fir were used significantly more frequently than lodgepole pine (32% and 26% compared to 7%). Among large wildlife trees, spruce was used significantly more frequently than lodgepole pine (70% compared to 29%), but the rate of use of subalpine fir (39%) was not significantly different from either species. No small wildlife trees were recently used by Pileated Woodpeckers compared to 10.4% of large wildlife trees.

After a stand-replacing fire, the density of wildlife trees will be approximately equal to the total tree density that existed before the fire. Over time these trees will begin to fall, so that eventually, none of them will remain standing. Within stands between 63 and 170 years of age, most trees killed in the stand-initiating fires had already fallen over and were counted only as CWD. No samples were available from stands between ages of 0 and 63 years (which is when almost all of these trees fall over), so that a survival function of standing dead trees after a fire could not be estimated. Only two small wildlife trees of 366 sampled (0.5%) could be identified as having died in the stand-initiating fire and still standing at the time of sampling between 63 and 170 years post-fire. Large wildlife trees from the pre-fire stand accounted for a larger percentage of the total number of large wildlife trees sampled (7 of 70 sampled; 10.0%), since (as found in section 2) they tended to stand longer after being killed by fires than did small trees.

Development of small wildlife trees (Figure 20A) from within the post-fire cohort was very rapid, with densities reaching up to 600/ha as early as 80 years after fire, and maximum levels of 1100/ha at stand ages of between 125 and 140 years. There was no significant correlation with stand age (r^2=0.01, $F_{1,48}$=0.71, P=0.40). Development of large wildlife trees from within the post-fire cohort can only begin as trees reach large sizes. A significant threshold appeared at a stand age of 124 years, when large wildlife trees in riparian sites developed from within the post-fire cohort started to appear at densities higher than 10/ha (Figure 20B). For the main study, this age was very similar (125 years). In marked contrast to the main study however, wildlife trees developed significantly more quickly from within the riparian post-fire stands in the Echo/Alex drainages than in the upland stands ($F_{1,47}$=11.37, P=0.002). No sites were found in the upland sites in the Echo/Alex drainages with more than five large wildlife trees/ha, while 14 of 31 riparian stands had more than five large wildlife trees/ha.

7.4.7 Coarse woody debris

The CWD volumes developed from the trees of the pre-fire stand were a function of present stand age and apparently the size of the trees in the pre-fire stand, as found in section 2. However, not enough pre-fire stands could be reliably aged in the Echo/Alex drainages; consequently, this parameter could not be compared between studies. High volumes of CWD (75-235 m^3/ha) were present in some stands between 67 and 88 years post-fire, but this CWD tended to decay quickly, and very little remained in any stand 90 or more years post-fire (except in a very decayed state, or class 5, and was consequently not counted in our surveys). The negative correlation with stand age was significant (r^2=0.24, $F_{1,47}$=15.32, P=0.000, Figure 21A) in the Echo/Alex drainages, and was not significantly different from section 2 ($F_{1,80}$=2.35, P=0.13). Neither study had a significant topographic effect ($F_{1,47}$=0.20, P=0.66 for Echo/Alex).

Table 22. Riparian to upland comparisons of wildlife tree densities and use by biogeoclimatic zone in the Echo and Alex drainages. Figures that are significantly different from the main study are highlighted within boxes.

Parameter	MSdm1 young stands		ESSFxc young stands		ANOVA row-wise statistics[1]		Old-growth
	Streamside Mean S.E.	Upland Mean S.E.	Streamside Mean S.E.	Upland Mean S.E.	$F_{3,42}$	P	Mean S.E.
SMALL (<30 CM DBH) WILDLIFE TREES IN ECHO AND ALEX DRAINAGES							
(No. of sites)	(21)	(10)	(9)	(5)			(1)
Density	370	312	315	420	0.29	0.83	149
	67	59	77	74			
Density recently used	33.4	40.5	10.7	0	0.59	0.63	0
	18.2	24.7	10.7	-			
% recently used	9.2	9.9	3.7	0	0.85^2	0.47	0
	3.5	5.2	3.7	-			
SMALL (<30 CM DBH) WILDLIFE TREES IN THE MAIN STUDY (PART 2)							
(No. of sites)	(15)	(16)	(4)	(4)			(8)
Density	152	200	148	331			118
	18	44	52	174			23
Density recently used	25.4	51.3	39.1	32.8			6.3
	4.3	12.2	21.7	5.0			2.0
% recently used	17.5	29.3	23.8	18.1			4.3
	3.3	6.8	11.6	6.5			2.8

SMALL (<30 CM DBH) WILDLIFE TREE COMPARISON STATISTICS BETWEEN TWO STUDIES[4]

	t	P	t	P	t	P	t	P
Density	3.14	0.00	1.53	0.15	1.36	0.20	0.51	0.63
Density recently used	0.42	0.68	0.43	0.67	1.32	0.21	2.19	0.12
% recently used	1.67	0.11	2.06	0.053	2.18	0.052	2.80	0.07

LARGE (≥30 CM DBH) WILDLIFE TREES IN ECHO AND ALEX DRAINAGES							
	Mean S.E.	Mean S.E.	Mean S.E.	Mean S.E.	$F_{3,42}$	P	Mean S.E.
(No. of sites)	(21)	(10)	(9)	(5)			(1)
Density	12.9^a	3.0^b	4.4^b	0^b	6.01	**0.002**	35
	2.3	1.5	1.5	-			
Density recently used	4.5	2.0	0.6	0	2.38	0.08	25
	1.2	1.5	0.6	-			
% recently used	33.1	50.0	16.7	-	0.87^3	0.43	71.4
	8.4	28.9	16.7	-			
LARGE (≥30 CM DBH) WILDLIFE TREES IN MAIN STUDY (PART 2)							
(N sites)	(15)	(16)	(4)	(4)			(8)
Density	11.3	4.2	10.3	13.0			36.8
	2.4	1.7	4.6	4.2			5.4
Density recently used	7.1	2.8	0.5	3.7			6.5
	1.6	1.0	0.5	0.8			4.1
% recently used	46.3	37.2	2.2	43.5			19.6
	7.6	10.5	2.2	13.5			4.7

LARGE (≥30 CM DBH) WILDLIFE TREE COMPARISON STATISTICS BETWEEN TWO STUDIES[4]

	t	P	t	P	t	P	t	P
Density	0.46	0.65	0.49	0.63	1.56	0.15	3.11	0.053
Density recently used	1.11	0.27	0.13	0.90	0.06	0.95	-	-
% recently used	1.15	0.26	0.51	0.61	0.69	0.51	-	-

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[1] Row-wise means with like superscripts are not significantly different when row-wise P <0.05.

[2] F using 3 num. d.f. and 40 denom. d.f., since sites with no small wildlife trees were not included in % calculations.

[3] F using 2 num. d.f. and 24 denom. d.f., since sites with no large wildlife trees were not included in % calculations.

[4] *t*-test between cell-by-cell study comparisons were adjusted when variances were significantly non-homogeneous.

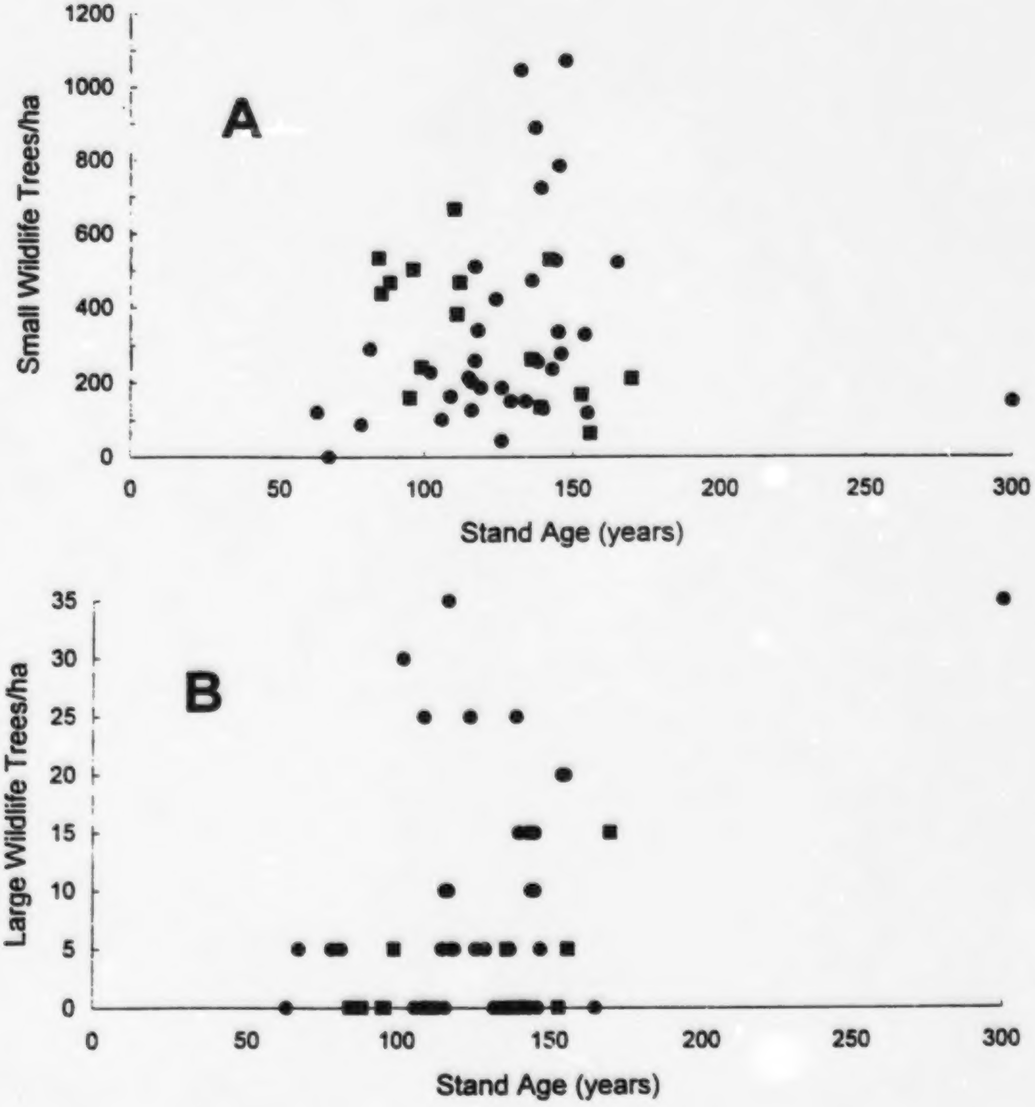

Figure 20. Density plotted against stand age for (A) <30 cm dbh and (B) ≥30 cm dbh wildlife trees in streamside and upland plots in the Echo and Alex drainages.

As in the main study, the volume of CWD developed from trees growing within the post-fire stand increased significantly with stand age in the Echo/Alex drainages (r^2=0.18, $F_{1,48}$=10.77, P=0.002, Figure 21B). No significant differences could be shown, using ANCOVA, between streamside and upland stands ($F_{1,47}$=0.14, P=0.71). The predicted mean was zero (i.e., no CWD developed from trees grown in the post-fire cohort) at 44 years post-fire and increased to 115 m^3/ha at 170 years, which was the practical limit of extrapolation in the Echo/Alex drainages because so few older stands were found. While these predictions appear to be slightly lower than in the main study, they were not significantly different ($F_{1,80}$=1.13, P=0.29).

7.4.8 Breeding birds

Comparisons could be made between 10 habitat types in the Echo/Alex and the main study (section 3) on breeding birds. The first major comparison was for eight habitats of any species with more than 35 total detections in the section 3 (Table 23). Major habitat types were clearcut and forest subdivided by biogeoclimatic zone and then by topographic position (streamside or upland) (Table 23). To keep habitats equivalent between studies, only those streams with forests on both sides were used for the riparian forest comparisons. Comparisons were not made for three species that were not detected or were uncommon during the Echo/Alex sampling because of the very late spring migration in 1995: Dusky Flycatchers, Willow Flycatchers, and Swainson's Thrushes.

Significant differences could be shown in at least one habitat class for 11 of the 22 species examined. Brown-headed Cowbirds were more abundant in ESSF riparian clearcuts in the Echo/Alex drainages. This probably does not reflect a real abundance, since the results may have been influenced by the presence of one large flock of Brown-headed Cowbirds, which are not territorial, and are therefore poorly sampled by traditional songbird counting methods. Differences for 5 of the 11 species were primarily within clearcuts — and were almost certainly related to the generally advanced state of regeneration in the clearcuts in the Echo/Alex drainages. In particular, Chipping Sparrows, Hermit Thrushes, Warbling Vireos, and Wilson's Warblers were more abundant in some of the clearcut habitats. The late-May sampling was also at the height of the Hermit Thrush and Wilson's Warbler migration period, so that some of the significantly higher detection densities for these species may be unrelated to habitat differences. Lincoln's Sparrows were less abundant than in the main study — and this is probably related to their preferences for open swampland, or recent and wet clearcuts, to wet clearcuts at a more advanced stage of regeneration.

Five of the 11 species were more abundant in some forest habitats in the Echo/Alex drainage than in the main study (Table 23). Ruby-crowned Kinglets were more abundant in all of the forested habitats in the Echo/Alex drainages than in the main study where they were very uncommon in all MSdm1 control forests and moderately common in ESSF forests. Only in streamside corridor forests did they become abundant in the main study. Red-breasted Nuthatches were more abundant in all MSdm1 forests and in ESSF riparian forests in the Echo/Alex drainages than in the main study, where they were only abundant in ESSF upland forests. Townsend's Warblers and Winter Wrens were more abundant in the MSdm1 upland forests in the Echo/Alex drainages than in the main study, where they were extremely uncommon in upland MSdm1 lodgepole pine forests. Yellow-rumped Warblers were more abundant in all the forest habitats in the Echo/Alex drainages, but particularly in the streamside MSdm1 forests, where they were least common in the main study. Total detection densities of all species was significantly higher in the streamside MSdm1 forests of the Echo/Alex drainages than in the main study.

The data from the Echo/Alex drainages were more closely examined to see why these differences may have occurred. Each transect that sampled 3 ha or more in total was considered a single data point so that means and variances could be calculated and various habitats compared statistically within the drainages. Ruby-crowned Kinglets were more than twice as common (0.65 vs. 0.28 and 0.23 birds/ha) in riparian forest edges (i.e., those riparian forests that formed wetland or lake boundaries, that were clearcut on one side of the stream, or were in streamside forest corridors) compared to streamside forests that spanned streams or to upland forests ($F_{2,26}$=9.46, P=0.001). However, even the lower of these abundances (i.e., 0.23/ha) were still higher than those in the main study. The overall high abundance of Ruby-crowned Kinglets in the Echo/Alex drainages may therefore have been a habitat response to forest fragmentation effects at the drainage level. The main study, which selected 5-ha controls relatively unaffected by fragmentation, may not have detected this effect.

Both Red-breasted Nuthatches and Yellow-rumped Warblers were simply more abundant in all the forests in the Echo/Alex drainages than in the forests of the main study. No correlation could be

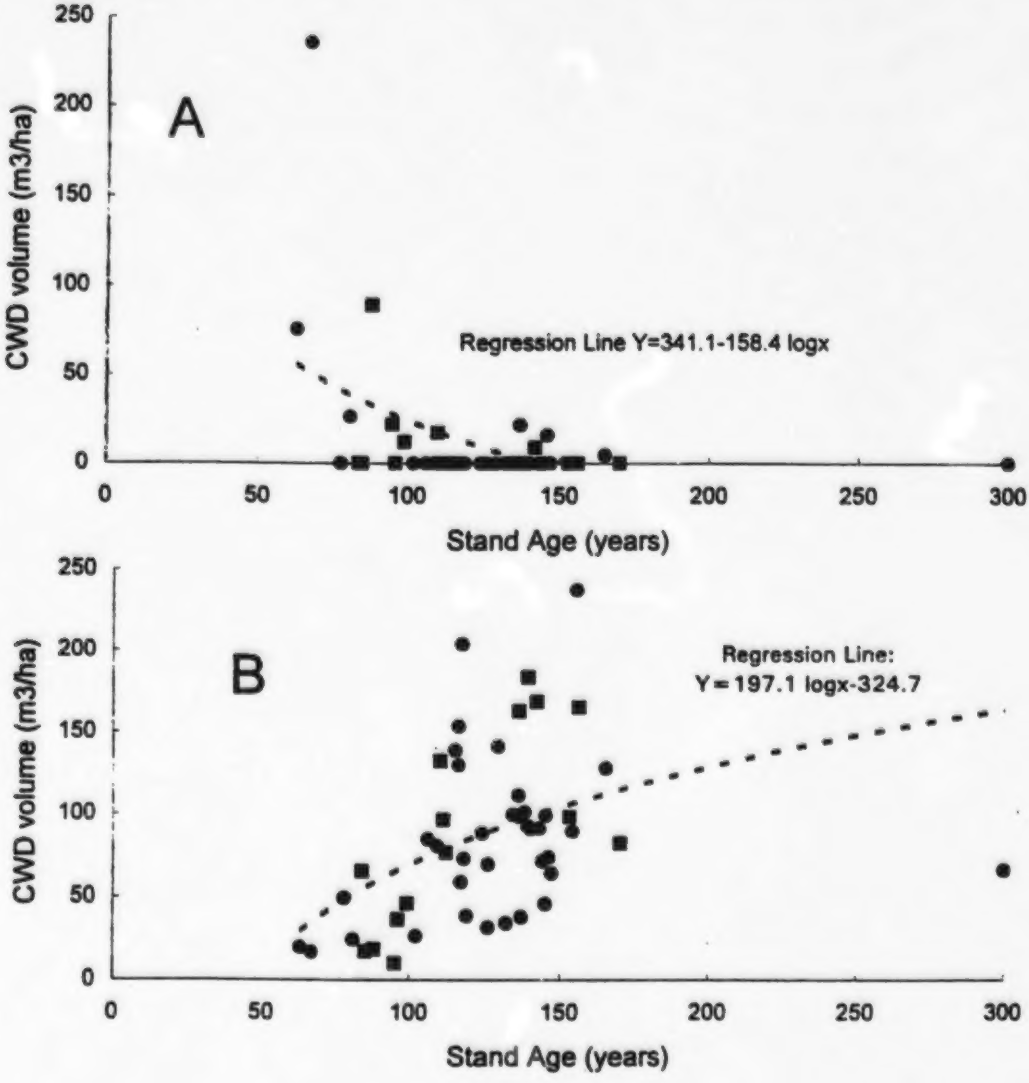

Figure 21. CWD volume developed from (A) the pre-fire, (B) the post-fire cohorts regressed against stand age for the Echo/Alex drainages in streamside and upland sites.

found with any particular habitat factors or landscape features within the drainages. For Yellow-rumped Warblers, there may have been some landscape-level response to fragmentation (since they were more abundant in forest corridors as wide as 188 m mean width in the main study), and the Echo/Alex forests are definitely fragmented to the point where few forests are more than 200 m from a clearcut edge. I cannot suggest at this time any particular factor that may have caused Red-breasted Nuthatches to be more abundant.

Winter Wrens were not found in the pure upland pine forests in the Echo/Alex drainages, just as in the main study. However, they were only detected on two of the upland transects in the Echo/Alex drainages. On one of the transects there were several wet depressions where veteran trees were abundant, and the site had more of the structural characteristics of old-growth forests than typical upland lodgepole pine forests. The other transect was a 136-year-old forest that was above 1500 m, which was the usual elevation boundary for the MSdm1/ESSFxc in the area. This northeastern corner of the Alex drainage had not been mapped as ESSFxc, probably because the overall extent of land above 1500 m was not judged to be enough to map this hill as ESSFxc at the 1:100 000 scale. However, the forests there did tend to have more characteristics of old-growth forests than did typical upland MSdm1 forests (130 canopy spruce/ha, and higher than average CWD of 163 m^3/ha). Overall, however, these results for Winter Wrens indicate that they will not be limited to riparian forests in the MSdm1, and that they may be found in the upland when conditions are right (i.e., something other than pure lodgepole pine stands).

Townsend's Warblers were four times more common in the MSdm1 upland in the Echo/Alex drainages than in the main study (Table 23). This was not an aberration of a few transects — the results were consistent across all the upland transects in the Echo/Alex drainages, with Townsend's Warblers found on every upland transect that sampled more than 2 ha. However, even in the MSdm1 of the Echo/Alex drainages, they were still more than twice as common in riparian forests than upland forests ($F_{1,12}$=51.59, P=0.000). Consequently, while Townsend's Warblers appeared almost limited to riparian forests in the MSdm1 in section 3, it appears that there are conditions under which they can be widespread (although still not as common as in riparian forests) in upland MSdm1 forests, such as in the Echo/Alex drainages.

At this point the author does not have any completely adequate explanation as to why Townsend's Warblers should be so widespread in the Echo/Alex forests, and absent or very uncommon in the upland MSdm1 control forests of the main study. There could be higher-level landscape effects, e.g., proximity to ESSF forests. Some of the control unharvested sites in the main study were >5 km from the nearest ESSF forests, so this may have had some effect. To verify any such effect would require extensive sampling to at least a reconnaissance level throughout the Okanagan Highlands.

In wide forest streamside corridors (mean width 188 m, range 132-280 m in section 3), no species group differed significantly in densities between the main study and the Echo/Alex drainages (Table 24). I could not compare narrower corridors in the Echo/Alex drainages because there were no narrow corridors. Streamside forests had either been left in wide corridors at streamsides, or had been cut right to the banks.

In wetlands, only two species were significantly more abundant than predicted at the 95% level in section 3 (Table 25). Savannah Sparrows and Spotted Sandpipers were significantly more abundant in the Echo/Alex drainages because more of the wetland samples in the Echo/Alex drainages came from sedge marshes that Savannah Sparrow prefer and the lakeshores that Spotted Sandpipers prefer. In the main study, swamps with more shrub cover were selected for study, in which Spotted Sandpipers would not find the open shoreline conditions that they prefer, and Savannah Sparrows would generally not find the sedge meadow habitat that they prefer.

7.4.9 Amphibians

Breeding pond searches for amphibians were not specifically undertaken in the Echo/Alex drainages, but all detections made during bird surveys were noted. Amphibians (Spotted Frog, Pacific Treefrog, and Western Toad) were noted breeding or calling at only eight locations in the two drainages. Four locations were in wetlands >2 ha and surrounded by forests, three locations were along streams that had been clearcut to the stream banks, and one was in a small depression in an upland clearcut. No amphibians were found breeding along streams that were completely forested or in upland forests. These results are similar to the results from the main study (section 5), where breeding in wetlands appeared relatively

Table 23. **Density of bird detections by habitat (forest and clearcut), biogeoclimatic zone, and topographic position in the Echo and Alex drainages compared to the main study for species with >35 detections in the main study.** Detection densities that are significantly different from the main study are highlighted within boxes.

Species	No. of birds		Density of detections (no. per ha)							
			Clearcut				Forest			
			MS		ESSF		MS		ESSF	
			Riparian	Upland	Riparian	Upland	Riparian	Upland	Riparian	Upland
American	111	Mean	0.180	0.207	0.213	0.120	0	0	0.007	0.013
Robin		S.D.	0.259	0.153	0.154	0.096			0.015	0.018
	20	Echo/Alex	0.421	0.121	0	0.134	0	0	0	0
Brown-headed	57	Mean	0.173	0.033	0.133	0.040	0	0	0.007	0.013
Cowbird		S.D.	0.171	0.033	0.217	0.089			0.015	0.018
	28	Echo/Alex	0.211	0.030	1.550	0.067	0.069	0.022	0	0
Chipping	99	Mean	0.093	0.140	0.127	0.193	0	0	0.020	0.087
Sparrow		S.D.	0.049	0.072	0.150	0.198			0.045	0.107
	33	Echo/Alex	0.253	0.333	0.097	0.403	0	0.043	0	0
Dark-eyed	424	Mean	0.733	0.573	0.333	0.600	0.100	0.127	0.200	0.160
Junco		S.D.	0.415	0.234	0.118	0.297	0.085	0.064	0.062	0.064
	59	Echo/Alex	0.589	0.242	0.388	0.403	0.092	0.087	0.050	0.233
Evening	187	Mean	0.007	0	0.240	0.173	0.020	0.080	0.300	0.427
Grosbeak		S.D.	0.015		0.316	0.220	0.030	0.056	0.364	0.530
	14	Echo/Alex	0.042	0	0.097	0	0.046	0	0.174	0
Golden-	296	Mean	0	0	0.007	0	0.607	0.287	0.453	0.620
crowned		S.D.			0.015		0.095	0.226	0.311	0.157
Kinglet	114	Echo/Alex	0	0	0	0	0.830	0.478	0.595	0.505
Gray Jay	47	Mean	0	0	0.053	0.073	0.020	0.080	0.007	0.080
		S.D.			0.056	0.164	0.030	0.077	0.015	0.061
	14	Echo/Alex	0.042	0.091	0	0.202	0.081	0	0	0.078
Hermit Thrush	78	Mean	0.033	0.020	0.113	0.040	0.047	0.007	0.160	0.100
		S.D.	0.047	0.030	0.115	0.060	0.056	0.015	0.055	0.053
	37	Echo/Alex	0.253	0.212	0.388	0.235	0.046	0.087	0.074	0.078
Lincoln's	167	Mean	0.580	0.133	0.380	0.020	0	0	0	0
Sparrow		S.D.	0.119	0.097	0.266	0.018				
	17	Echo/Alex	0.211	0.182	0.194	0.067	0	0	0	0
Northern	63	Mean	0.327	0	0.013	0	0.020	0.020	0.033	0.007
Waterthrush		S.D.	0.675	-	0.018	-	0.030	0.045	0.075	0.015
	26	Echo/Alex	0.042	0.030	0.097	0	0.194	0	0.102	0.039
Orange-	71	Mean	0.113	0.213	0.080	0.047	0	0.007	0	0.013
crowned		S.D.	0.164	0.135	0.084	0.104		0.015		0.018
Warbler	9	Echo/Alex	0.211	0.061	0.097	0.034	0	0	0	0
Pine Siskin	389	Mean	0.200	0.160	0.353	0.340	0.113	0.327	0.473	0.627
		S.D.	0.178	0.189	0.281	0.317	0.080	0.098	0.344	0.229
	7	Echo/Alex	0.042	0	0	0	0	0.109	0.025	0
Red-breasted	38	Mean	0	0	0.013	0.007	0.020	0.033	0.007	0.173
Nuthatch		S.D.			0.030	0.015	0.030	0.024	0.015	0.119
	34	Echo/Alex	0	0	0	0.067	0.230	0.299	0.074	0.078

Table 23 (continued)

Species	No. of birds		Density of detections (no. per ha)							
			Clearcut				Forest			
			MS		ESSF		MS		ESSF	
			Riparian	Upland	Riparian	Upland	Riparian	Upland	Riparian	Upland
Ruby-crowned Kinglet	48	Mean	0.040	0	0.127	0.013	0.020	0.007	0.047	0.067
		S.D.	0.089		0.159	0.030	0.018	0.015	0.069	0.067
	62	Echo/Alex	0.211	0	0	0.034	0.277	0.217	0.198	0.233
Red Crossbill	56	Mean	0	0.020	0	0	0.033	0.207	0.007	0.107
		S.D.		0.045			0.047	0.231	0.015	0.104
	1	Echo/Alex	0	0	0	0	0	0	0	0
Townsend's Warbler	276	Mean	0	0	0	0	0.620	0.080	0.560	0.580
		S.D.					0.353	0.065	0.344	0.333
	100	Echo/Alex	0	0.030	0	0	0.807	0.326	0.595	0.388
Tree Swallow	36	Mean	0.053	0.080	0.080	0.027	0	0	0	0
		S.D.	0.030	0.179	0.102	0.060				
	27	Echo/Alex	0.042	0.121	0	0	0	0	0	0
Warbling Vireo	80	Mean	0.133	0.240	0.127	0.033	0	0	0	0
		S.D.	0.097	0.292	0.161	0.058				
	15	Echo/Alex	0.463	0.030	0.194	0	0	0	0.025	0
White-crowned Sparrow	148	Mean	0.227	0.180	0.373	0.207	0	0	0	0
		S.D.	0.132	0.177	0.312	0.175				
	6	Echo/Alex	0.126	0.061	0	0.034	0	0	0	0
Wilson's Warbler	133	Mean	0.427	0.020	0.267	0.060	0.020	0.013	0.027	0.053
		S.D.	0.514	0.045	0.176	0.083	0.045	0.030	0.043	0.087
	33	Echo/Alex	0.716	0.152	0	0.067	0	0	0.198	0
Winter Wren	70	Mean	0.020	0	0	0.007	0.213	0	0.173	0.053
		S.D.	0.045			0.015	0.096		0.114	0.038
	13	Echo/Alex	0	0	0.097	0	0.115	0.087	0	0
Yellow-rumped Warbler	181	Mean	0.107	0	0.167	0.053	0.127	0.340	0.187	0.227
		S.D.	0.136		0.103	0.087	0.080	0.098	0.090	0.126
	108	Echo/Alex	0.379	0.061	0.291	0.067	0.715	0.435	0.273	0.427
Total all species	3612	Mean	4.400	2.527	3.887	2.320	2.207	1.793	2.913	4.033
		S.D.	1.473	1.059	1.868	0.702	0.513	0.377	1.519	0.999
	899	Echo/Alex	5.390	2.091	4.175	1.983	3.919	2.435	2.827	2.447
Area sampled (ha)		Main study	5 5-ha plots visited six times each for each mean (equivalent to 150 ha sampled once)							
		Echo/Alex	23.75	33.00	10.30	29.75	43.38	46.00	40.33	25.75

Table 24. Density of bird species groups detections in wide (132-280 m) streamside forest corridors in the Echo and Alex drainages compared to the main study.

	Bird detections/ha in wide streamside forest corridors		
	Main study		Echo/Alex
Species group	Mean	S.D.	mean
Total density	4.16	0.65	3.10
Width-sensitive forest species[1]	0.87	0.04	0.93
Other forest species[2]	0.65	0.38	0.78
Open-forest species[3]	1.63	0.40	1.32
Generalist species[4]	0.63	0.25	0
Other species	0.36	0.34	0.16
Area sampled	5 5-ha plots visited 6 times each (area equivalent to 150 ha)		12.9 ha

[1] Brown Creeper, Gray Jay, Townsend's Warbler, and Winter Wren.

[2] Evening Grosbeak, Golden-crowned Kinglet, Red-breasted Nuthatch, Red Crossbill, Three-toed Woodpecker, and Varied Thrush.

[3] Fox Sparrow, Hammond's Flycatcher, Hermit Thrush, Mountain Chickadee, Northern Waterthrush, Olive-sided Flycatcher, Ruby-crowned Kinglet, Swainson's Thrush, Wilson's Warbler, and Yellow-rumped Warbler.

[4] Dark-eyed Junco and Pine Siskin.

unaffected by clearcutting, but clearcutting opened up breeding possibilities in very small pools that were too cold for breeding under a closed forest canopy.

7.5 FPC Riparian Guideline Application

The recommendations of the FPC guidelines were applied to the Echo/Alex drainages to calculate the area affected and determine potential effects on wildlife habitat and timber-harvest levels. Both drainages were assumed not to be in a community watershed (although they are both in the Vernon Creek community watershed), so that the most stringent guidelines would not necessarily apply.

In total, the effects of applying the FPC riparian guidelines on timber-harvest levels were minor (Table 26). Total reserve zones around S3 streams, W1 wetlands, and L1 and L2 lakes only applied to 22.53 ha, which amounted to only 1.20% of the forested area in the Echo drainage and 0.98% in the Alex drainage. Additional timber retention in RMZs accounted for an additional equivalent of 1.36% of the forested area in the Alex drainage and 3.85% in the Echo drainage.

The Echo drainage turned out to have a higher level of timber retention because of the presence of Echo Lake. The FPC riparian guidelines do not designate management zones around L1 lakes; that is left up to the district manager. In this case, the Okanagan Timber Supply Area (TSA) guidelines (Ministry of Forests 1992) were grandfathered into the lakeshore management guidelines. Under the Okanagan TSA guidelines, Echo Lake was a Class A lake with a 200-m wide lakeshore management zone within which there will be no clearcutting or road-building, and a maximum of 50% canopy reduction through partial cutting. This lakeshore management zone alone accounted for 2.91% of the additional 3.85% total timber retention in the Echo drainage.

It was intended to measure the extent of riparian forest in the drainages and indicate whether the guidelines would actually protect these riparian forests. Unfortunately, 1:15 000-scale mapping proved to be too coarse a scale to map the riparian forest accurately in the area, since riparian forests were usually too narrow to be mappable at this scale. However, most of the riparian forests around the FPC-classified streams, wetlands, and lakes were contained within riparian reserve and management zones. However, there were also many shallow gullies with intermittent streams, and shallow depressions that interconnected to form drainage features, but had no stream channels that had the spruce-dominant characteristics of riparian sites. These gullies and depressions often had hummocky organic soils, but will not receive any recognition under the FPC riparian guidelines. It was not possible to identify what percentage of area these features covered without further ground-truthing. Intensive mapping at 1:5000 scale would be required to do this sort of mapping at the site series level.

Table 25. Density of wetland bird detections in the Echo and Alex drainages compared to the main study. Detection densities that are significantly different from the main study are highlighted within boxes.

	Bird detections/ha in wetlands		
	Main breeding bird study		Echo/Alex drainages
Species	Mean	S.D.	Mean
American Robin	0.104	0.138	0.104
Brown-headed Cowbird	0.200	0.149	0.208
Common Snipe	0.170	0.138	0.156
Common Yellowthroat	1.067	0.832	0
Dark-eyed Junco	0.163	0.106	0.052
Gray Jay	0.030	0.089	0.156
Lincoln's Sparrow	0.681	0.244	0.573
Mallard	0.511	0.747	0.417
Northern Waterthrush	0.385	0.235	0.469
Orange-crowned Warbler	0.141	0.113	0.260
Pine Siskin	0.178	0.149	0
Ruby-crowned Kinglet	0.133	0.133	0
Ring-necked Duck	0.111	0.333	0
Red-winged Blackbird	0.496	0.948	0
Savannah Sparrow	0.037	0.048	$\boxed{0.313}$
Solitary Sandpiper	0.081	0.182	0.156
Song Sparrow	1.289	0.534	0.625
Spotted Sandpiper	0.007	0.022	$\boxed{0.365}$
Tree Swallow	0.541	0.993	0.156
Warbling Vireo	0.237	0.256	0.052
Willow Flycatcher	0.904	0.629	0
Wilson's Warbler	0.526	0.396	0.990
Yellow Warbler	0.363	0.513	0
Yellow-rumped Warbler	0.096	0.106	0
Total all species	9.207	3.588	5.158
Area sampled	Nine 2.5-ha plots visited six times each (equivalent to 135 ha sampled once)		19.195 ha

It was intended to make a simple model to project the effects of FPC riparian guidelines on certain key species of wildlife, but there simply was not the data to do so with any reliability. The simplest model would be one of habitat suitability immediately after timber harvest, which can be made based on the stand-level data of the main study with a fair degree of reliability. These results have already been discussed in sections 2, 3, and 5 of this report. However, entire drainages will not be harvested all at once, so that projection of habitat suitability into the managed stands of the future will be essential to understanding how wildlife populations will respond at levels larger than the stand.

Unfortunately, reliable data are lacking to make such long-term projections. For example, Winter Wrens and Townsend's Warblers require, or were most abundant in, riparian forests in the MSdm1. They need wider forest corridors than are likely to be provided for in reserve zones in any of the streams in the Echo/Alex drainages. Therefore, immediately after timber harvesting, these riparian reserve strips will be unsuitable habitat. An exception might be along S3 streams, where a 40-m total reserve width (including both sides of the stream) may be suitable if the additional 20 m on each side of the reserved zone is

Table 26. Effects on timber-harvest levels in the Echo and Alex drainages when the Forest Practices Code riparian timber-harvest guidelines are applied.

FPC riparian classification	Stream or perimeter length (m)	Reserve zone area (ha)	Management zone area (ha)	Maximum % timber retention in MZ	Total equivalent area netted out at 100% (ha)	% of total forested area
ECHO CREEK DRAINAGE, 881 FORESTED HA						
S3	1860	7.44	7.44	50	11.16	1.27
S4	1578	0	9.47	25	2.37	0.27
S6	2200	0	8.80	5	0.44	0.05
W1	260	0.26	1.04	25	0.52	0.06
W3	1925	0	5.78	25	1.45	0.16
L1 Class A lake	2897	2.90	51.26	50	28.53	3.24
				Total	44.47	5.05
	Actual total allowing for areas already clearcut to riparian edges				43.95	4.99
ALEX CREEK DRAINAGE, 1223 FORESTED HA						
S3	2544	10.18	10.18	50	15.27	1.25
S4	2940	0	17.64	25	4.41	0.36
S6	8036	0	32.14	5	1.61	0.13
W3	6283	0	18.85	25	4.71	0.39
L1 Class B lake	980	0.98	1.96	25	1.47	0.12
L2 Class C lake	772	0.77	1.54	25	1.16	0.09
				Total	28.63	2.34
	Actual total allowing for areas already clearcut to riparian edges				25.02	2.05

retained at high-retention levels. Even then, the total width of the riparian-zone forest will only be 80 m, which is just at the limit of suitability for these species. It is likely that taking 50% of the timber out of the outermost parts of the zone will not be equivalent to a partial cut that takes only 25% out of the whole zone, because 50% partial harvest levels (in lodgepole pine stands) near to some of the main riparian study sites were unused by Townsend's Warblers.

As the managed forest regenerates around riparian reserve zones, both Townsend's Warblers and Winter Wrens are likely to re-inhabit these riparian forests when the surrounding forest regains canopy height and closure (probably after 40-50 years). However, reserve corridors occupy only a very small portion of the stream lengths. Whether these species will return to clearcut riparian management zones (with 5-25% retention of trees at time of harvest) 40-50 years after timber harvest will depend on canopy closure and tree species composition for Townsend's Warbler, as well as CWD levels for Winter Wrens.

Once timber harvesting actually begins in cutblocks laid out under the FPC riparian guidelines, then there will be opportunity to re-examine the assumptions of the guidelines and see what the actual short-term effects are on wildlife, trees, and structural attributes. At that time, reconnaissance-level sampling similar to that conducted within this internal validation study should be able to quickly assess the actual effects of the guidelines, using the main study and the Echo/Alex data as baseline data.

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9 APPENDICES

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Appendix A. **Mean numbers of detections for common bird species and for the total abundance of birds detected within the MS–ESSF streamside breeding bird investigation. Numbers detected are the sums of detections on six visits to 5-ha plots. See Appendix I for scientific names of birds listed in this report.**

	No. detected in 5-ha plots visited six times each (N=5 for each cell)									
	Montane Spruce zone					Engelmann Spruce-Subalpine Fir zone				
		Topographic position (TP)					Topographic position (TP)			
		Riparian		Upland			Riparian		Upland	
COMMUNITY MACROPARAMETER	Forest treatment	Mean	S.E.	Mean	S.E.	Forest treatment	Mean	S.E.	Mean	S.E.
Total abundance	Control	66.2	6.9	53.8	5.1	Control	87.4	20.4	121.0	13.4
	Corridor	153.0	9.6	68.8	4.8	Corridor	97.4	8.5	123.2	18.2
	Clearcut	132.0	19.8	75.6	14.2	Clearcut	116.6	25.1	69.6	9.4
Species richness	Control	12.2	1.5	11.2	1.3	Control	13.0	1.1	18.6	1.3
(No. species per plot)	Corridor	25.0	1.7	15.2	1.7	Corridor	16.6	2.2	19.6	2.3
	Clearcut	22.4	1.5	15.4	1.6	Clearcut	20.8	2.8	12.2	1.5
SPECIES										
American Robin	Control	0	-	0	-	Control	0.2	0.2	0.4	0.2
	Corridor	3.8	1.1	2.8	1.6	Corridor	1.8	0.9	0	-
	Clearcut	5.4	3.5	6.2	2.1	Clearcut	6.4	2.1	3.6	1.3
Boreal Chickadee	Control	0	-	0	-	Control	0.8	0.6	6.0	3.5
	Corridor	0	-	0	-	Corridor	0	-	1.2	0.8
	Clearcut	0	-	0	-	Clearcut	0	-	0	-
Brown-headed Cowbird	Control	0	-	0	-	Control	0	-	0	-
	Corridor	5.4	1.2	0.6	0.2	Corridor	0.4	0.4	0.6	0.6
	Clearcut	5.2	2.3	1.0	0.4	Clearcut	4.0	2.9	1.2	1.2
Brown Creeper	Control	1.6	0.8	1.8	1.3	Control	0.4	0.2	2.4	1.0
	Corridor	2.0	0.9	0.4	0.2	Corridor	0.6	0.4	1.0	0.4
	Clearcut	0	-	0	-	Clearcut	0	-	0	-
Chipping Sparrow	Control	0	-	0	-	Control	0.6	0.6	2.6	1.4
	Corridor	1.3	1.3	0.6	0.4	Corridor	0.8	0.5	0.4	0.2
	Clearcut	2.8	0.7	4.2	1.0	Clearcut	3.8	2.0	5.6	2.7
Dark-eyed Junco	Control	3.0	1.1	3.8	0.9	Control	6.0	0.8	4.8	0.9
	Corridor	21.6	3.8	11.8	3.0	Corridor	10.8	1.7	3.6	1.3
	Clearcut	22.0	5.6	17.2	3.1	Clearcut	10.0	1.6	18.0	4.0
Dusky Flycatcher	Control	0	-	0	-	Control	0	-	0	-
	Corridor	0.6	0.4	0	-	Corridor	0.2	0.2	0	-
	Clearcut	1.0	0.5	4.2	1.5	Clearcut	0.2	0.2	3.0	2.8
Evening Grosbeak	Control	0.6	0.4	2.4	0.7	Control	9.0	4.9	12.8	7.1
	Corridor	1.2	0.6	3.0	1.4	Corridor	1.2	1.2	0.6	0.4
	Clearcut	0.2	0.2	0	-	Clearcut	7.2	4.2	5.2	3.1
Golden-crowned Kinglet	Control	18.2	1.3	8.6	3.0	Control	13.6	4.2	18.6	2.1
	Corridor	15.2	1.2	7.6	2.4	Corridor	11.4	3.2	20.0	2.8
	Clearcut	0	-	0	-	Clearcut	0.2	0.2	0	-
Gray Jay	Control	0.6	0.4	2.4	1.0	Control	0.2	0.2	2.4	0.8
	Corridor	2.4	1.0	0.8	0.6	Corridor	0.6	0.4	2.4	1.7
	Clearcut	0	-	0	-	Clearcut	1.6	0.7	2.2	2.2
Hammond's Flycatcher	Control	0.8	0.6	0	-	Control	0.6	0.6	0	-
	Corridor	3.2	2.5	0	-	Corridor	1.2	1.2	0	-
	Clearcut	0	-	0	-	Clearcut	0	-	0	-
Hermit Thrush	Control	1.4	0.7	0.2	0.2	Control	4.8	0.7	3.0	0.7
	Corridor	3.8	1.0	1.8	0.9	Corridor	3.6	0.9	3.4	1.1
	Clearcut	1.0	0.6	0.6	0.4	Clearcut	3.4	1.5	1.2	0.8
Lincoln's Sparrow	Control	0	-	0	-	Control	0	-	0	-
	Corridor	0	-	0	-	Corridor	2.2	0.7	0	-
	Clearcut	17.4	1.6	4.0	1.3	Clearcut	11.4	3.6	0.6	0.2

Appendix A (continued)

	No. detected in 5-ha plots visited six times each (N=5 for each cell)									
	Montane Spruce zone					Engelmann Spruce-Subalpine Fir zone				
		Topographic position (TP)					Topographic position (TP)			
	Forest	Riparian		Upland		Forest	Riparian		Upland	
SPECIES	treatment	Mean	S.E.	Mean	S.E.	treatment	Mean	S.E.	Mean	S.E.
MacGillivray's Warbler	Control	0.4	0.2	0.4	0.4	Control	0	-	0	-
	Corridor	2.4	0.5	0.2	0.2	Corridor	0	-	1.4	0.7
	Clearcut	2.4	1.0	1.0	0.3	Clearcut	2.4	1.2	0	-
Mountain Bluebird	Control	0	-	0	-	Control	0	-	0	-
	Corridor	0	-	0	-	Corridor	0	-	0	-
	Clearcut	1.8	1.1	0.6	0.4	Clearcut	0.4	0.2	1.2	1.2
Northern Flicker	Control	0	-	0	-	Control	0	-	0	-
	Corridor	0.8	0.4	0.2	0.2	Corridor	0	-	0	-
	Clearcut	1.0	0.5	0.6	0.4	Clearcut	1.4	0.9	0.6	0.4
Northern Waterthrush	Control	0.6	0.4	0.6	0.6	Control	1.0	1.0	0.2	0.2
	Corridor	1.0	0.5	1.0	0.8	Corridor	1.4	0.7	0	-
	Clearcut	9.8	9.1	0	-	Clearcut	0.4	0.2	0	-
Orange-crowned Warbler	Control	0	-	0.2	0.2	Control	0	-	0.4	0.2
	Corridor	3.0	1.5	1.4	1.4	Corridor	0	-	1.2	0.8
	Clearcut	3.4	2.2	6.4	1.8	Clearcut	2.4	1.1	1.4	1.4
Pine Siskin	Control	3.4	1.1	9.8	1.5	Control	14.2	4.6	18.8	3.1
	Corridor	12.4	4.2	3.8	1.9	Corridor	10.6	1.9	9.8	3.2
	Clearcut	6.0	2.4	4.8	2.5	Clearcut	10.6	3.8	10.2	4.2
Red-breasted Nuthatch	Control	0.6	0.4	1.0	0.3	Control	0.2	0.2	5.2	1.6
	Corridor	4.4	1.1	0.6	0.2	Corridor	1.0	0.8	7.4	1.7
	Clearcut	0	-	0	-	Clearcut	0.4	0.4	0.2	0.2
Red Crossbill	Control	1.0	0.6	6.2	3.1	Control	0.2	0.2	3.2	1.3
	Corridor	0.8	0.5	2.6	0.5	Corridor	0	-	2.0	0.9
	Clearcut	0	-	0.6	0.6	Clearcut	0	-	0	-
Ruby-crowned Kinglet	Control	0.6	0.2	0.2	0.2	Control	1.4	0.9	2.0	0.9
	Corridor	8.0	0.9	3.0	1.8	Corridor	4.6	2.9	14.0	1.3
	Clearcut	1.2	1.2	0	-	Clearcut	3.8	2.1	0.4	0.4
Savannah Sparrow	Control	0	-	0	-	Control	0	-	0	-
	Corridor	0	-	0	-	Corridor	0	-	0	-
	Clearcut	1.8	1.1	0.8	0.6	Clearcut	4.4	3.7	0	-
Song Sparrow	Control	0	-	0	-	Control	0	-	0	-
	Corridor	0	-	0	-	Corridor	0.4	0.4	0	-
	Clearcut	5.2	2.7	0	-	Clearcut	1.2	0.6	0	-
Swainson's Thrush	Control	2.4	1.1	1.8	0.9	Control	2.0	0.5	3.6	0.5
	Corridor	3.2	1.2	2.6	0.7	Corridor	3.0	1.1	4.8	0.7
	Clearcut	0.8	0.6	0	-	Clearcut	0.2	0.2	0	-
Townsend's Warbler	Control	18.6	4.7	2.4	0.9	Control	16.8	4.6	17.4	4.5
	Corridor	18.6	2.2	1.6	1.1	Corridor	5.0	2.9	13.2	2.6
	Clearcut	0	-	0	-	Clearcut	0	-	0	-
Varied Thrush	Control	0.6	0.4	0	-	Control	3.4	0.7	1.6	0.7
	Corridor	0.8	0.6	0	-	Corridor	0.2	0.2	4.4	3.2
	Clearcut	0	-	0	-	Clearcut	0	-	0	-
Warbling Vireo	Control	0	-	0	-	Control	0	-	0	-
	Corridor	1.8	1.6	0.2	0.2	Corridor	0.2	0.2	2.8	2.1
	Clearcut	4.0	1.3	7.2	3.9	Clearcut	3.8	2.2	1.0	0.8
White-crowned Sparrow	Control	0	-	0	-	Control	0	-	0	-
	Corridor	0.6	0.4	0	-	Corridor	1.6	1.1	0.4	0.2
	Clearcut	6.8	1.8	5.4	2.4	Clearcut	11.2	4.2	6.2	2.4

Appendix A (continued)

	No. detected in 5-ha plots visited 6 times each (N=5 for each cell)									
	Montane Spruce zone					Engelmann Spruce-Subalpine Fir zone				
		Topographic position (TP)					Topographic position (TP)			
	Forest	Riparian		Upland		Forest	Riparian		Upland	
SPECIES	treatment	Mean	S.E.	Mean	S.E.	treatment	Mean	S.E.	Mean	S.E.
Willow Flycatcher	Control	0	-	0	-	Control	0	-	0	-
	Corridor	0	-	0	-	Corridor	0	-	0	-
	Clearcut	6.0	3.2	0.4	0.2	Clearcut	1.6	1.2	0	-
Wilson's Warbler	Control	0.6	0.6	0.4	0.4	Control	0.8	0.6	1.6	1.2
	Corridor	6.2	2.5	0.6	0.6	Corridor	14.4	4.2	7.0	4.3
	Clearcut	12.8	6.9	0.6	0.6	Clearcut	8.0	2.4	1.8	1.1
Winter Wren	Control	6.4	1.3	0	-	Control	5.2	1.5	1.6	0.5
	Corridor	4.8	1.5	0	-	Corridor	4.8	2.3	0.6	0.4
	Clearcut	0.6	0.6	0	-	Clearcut	0	-	0.2	0.2
Yellow-rumped Warbler	Control	3.8	1.1	10.2	1.3	Control	5.6	1.2	6.8	1.7
	Corridor	16.4	3.3	17.6	2.2	Corridor	11.4	2.2	12.0	2.1
	Clearcut	3.2	1.8	0	-	Clearcut	5.0	1.4	1.6	1.2

Appendix B. ANOVA statistics for common bird species and for the total abundance of birds detected in the MS–ESSF streamside breeding bird investigation. Data presented in Appendix A. BGC = biogeoclimatic zone effect; TP = topographic position effect; and FT = forest treatment effect. Effect size (f) of Cohen (1988), where 0.4 denotes large effect, 0.25 medium effect, and 0.1 small effect. Significant α denoted in **bold**.

3-factor ANOVA statistics					
Species Factors (* = interaction effect)	F	d.f. (num, denom.)	P (α)	Effect size (f)	Power (1-β)
Total abundance					
BGC*TP*FT	3.12	2,48	0.053	0.32	0.50
BGC*TP	10.92	1,48	**0.002**	0.43	0.85
BGC*FT	4.11	2,48	**0.022**	0.37	0.62
FT*TP	4.78	2,48	**0.013**	0.40	0.70
One-factor ANOVA	8.96	11,48	**.000**	1.28	1.00
153^{a} 132^{ab} 123^{ab} 121^{abc} 117^{abc} 97^{bcd} 87^{bcd} 76^{cd} 70^{d} 69^{d} 66^{d} 54^{d}					
American Robin					
BGC*TP*FT	0.43	2,48	0.65	0.12	0.11
BGC*TP	0.65	1,48	0.42	0.10	0.10
BGC*FT	0.82	2,48	0.45	0.17	0.17
FT*TP	0.27	2,48	0.77	0.09	0.08
BGC	1.25	1,48	0.27	0.14	0.16
TP	0.79	1,48	0.38	0.11	0.12
FT	12.55	2,48	**0.000**	0.65	0.99
Boreal Chickadee					
TP*FT	1.19	1,16	0.29	0.24	0.16
TP	3.04	1,16	0.10	0.39	0.34
FT	2.33	1,16	0.15	0.34	0.27

3-factor ANOVA statistics					
Species Factors (* = interaction effect)	F	d.f. (num, denom.)	P (α)	Effect size (f)	Power (1-β)
Chipping Sparrow					
BGC*TP*FT	1.14	2,48	0.91	0.19	0.20
BGC*TP	0.77	1,48	0.39	0.11	0.12
BGC*FT	1.11	2,48	0.34	0.19	0.20
FT*TP	1.28	2,48	0.29	0.21	0.24
BGC	1.14	1,48	0.29	0.14	0.16
TP	0.77	1,48	0.39	0.11	0.12
FT	10.15	2,48	**0.000**	0.58	0.96
Dark-eyed Junco					
BGC*TP*FT	1.90	2,48	0.16	0.25	0.32
BGC*TP	1.98	1,48	0.17	0.18	0.24
BGC*FT	1.03	2,48	**0.016**	0.39	0.67
FT*TP	3.83	2,48	**0.029**	0.36	0.60
BGC within Control	4.81	1,18	**0.04**	0.49	0.55
BGC within Corridor	8.72	1,18	**0.009**	0.66	0.80
BGC within Clearcut	2.01	1,18	0.17	0.32	0.27
FT within MS	12.40	2,27	**0.000**	0.91	0.99
FT within ESSF	7.09	2,27	**0.003**	0.69	0.90
TP within Control	0.04	1,18	0.85	0.04	0.05
TP within Corridor	6.36	1,18	**0.021**	0.56	0.66
TP within Clearcut	0.15	1,18	0.70	0.09	0.07
FT within Riparian	6.99	2,27	**0.004**	0.68	0.89
FT within Upland	13.85	2,27	**0.000**	0.96	1.00
Dusky Flycatcher					
BGC*TP*FT	0.06	1,32	0.81	0.04	0.06
BGC*TP	0.00	1,32	1.00	0	0.05
BGC*FT	0.24	1,32	0.62	0.08	0.07
TP*FT	4.41	1,32	**0.044**	0.33	0.46
BGC	0.55	1,32	0.46	0.12	0.10
TP within Corridor	3.27	1,18	0.09	0.40	0.40
TP within Clearcut	3.84	1,18	0.07	0.44	0.46
FT within Riparian	0.28	1,18	0.60	0.12	0.08
FT within Upland	5.76	1,18	**0.027**	0.54	0.63

Appendix B (continued)

	3-factor ANOVA statistics				
Species Factors	F	d.f.	P (α)	Effect size (f)	Power (1-β)
Brown-headed Cowbird					
BGC*TP*FT	0.75	1,32	0.39	0.14	0.12
BGC*TP	2.36	1,32	0.14	0.24	0.27
BGC*FT	0.92	1,32	0.34	0.15	0.14
FT*TP	0.33	1,32	0.57	0.09	0.06
BGC	2.08	1,32	0.16	0.23	0.25
TP	7.77	1,32	**0.009**	0.44	0.70
FT	1.12	1,32	0.30	0.17	0.16
Brown Creeper					
BGC*TP*FT	0.95	1,32	0.34	0.15	0.14
BGC*TP	2.26	1,32	0.14	0.24	0.27
BGC*FT	1.32	1,32	0.26	0.18	0.17
FT*TP	0.20	1,32	0.66	0.07	0.07
BGC	0.63	1,32	0.43	0.13	0.11
TP	0.38	1,32	0.54	0.10	0.09
FT	0.01	1,32	0.93	0.02	0.05
Gray Jay					
BGC*TP*FT	0.72	2,48	0.49	0.15	0.14
BGC*TP	1.65	1,48	0.21	0.17	0.22
BGC*FT	1.43	2,48	0.25	0.22	0.26
FT*TP	1.11	2,48	0.34	0.19	0.20
BGC	0.87	1,48	0.36	0.12	0.13
TP	1.96	1,48	0.17	0.18	0.24
FT	0.40	2,48	0.67	0.12	0.11
Hammond's Flycatcher					
BGC*FT	0.38	1,16	0.55	0.14	0.09
BGC	0.57	1,16	0.46	0.17	0.10
FT	1.06	1,16	0.32	0.23	0.15
Hermit Thrush					
BGC*TP*FT	1.15	2,48	0.33	0.20	0.22
BGC*TP	0.04	1,48	0.84	0.03	0.05
BGC*FT	2.04	2,48	0.14	0.26	0.34
FT*TP	0.05	2,48	0.95	0.04	0.06
BGC	12.77	1,48	**0.001**	0.46	0.89
TP	6.91	1,48	**0.012**	0.34	0.65
FT	3.49	1,48	**0.038**	0.24	0.55

	3-factor ANOVA statistics				
Species Factors	F	d.f.	P (α)	Effect size (f)	Power (1-β)
Evening Grosbeak					
BGC*TP*FT	0.16	2,48	0.85	0.07	0.07
BGC*TP	0.05	1,48	0.83	0.03	0.05
BGC*FT	3.32	2,48	**0.045**	0.33	0.52
FT*TP	0.43	2,48	0.65	0.12	0.11
TP	0.20	1,48	0.66	0.06	0.07
BGC within Control	5.15	1,18	**0.036**	0.51	0.58
BGC within Corridor	1.48	1,18	0.24	0.27	0.21
BGC within Clearcut	6.00	1,18	**0.025**	0.55	0.64
FT within MS	3.63	2,27	**0.040**	0.24	0.18
FT within ESSF	3.20	2,27	0.06	0.46	0.56
Golden-crowned Kinglet					
BGC*TP*FT	0.04	1,32	0.83	0.03	0.05
BGC*TP	16.50	1,32	**0.000**	0.64	0.95
BGC*FT	0.18	1,32	0.68	0.07	0.07
TP*FT	0.55	1,32	0.47	0.12	0.10
FT	0.40	1,32	0.53	0.10	0.09
BGC within Riparian	2.48	1,18	0.13	0.35	0.32
BGC within Upland	20.49	1,18	**0.000**	1.01	0.99
TP within MS	17.56	1,18	**0.001**	0.94	0.98
TP within ESSF	5.12	1,18	**0.036**	0.51	0.58
Northern Flicker					
BGC*TP*FT	0.64	1,32	0.43	0.13	0.11
BGC*TP	0.03	1,32	0.87	0.03	0.05
BGC*FT	1.26	1,32	0.27	0.18	0.17
FT*TP	0.23	1,32	0.63	0.08	0.07
BGC	0.23	1,32	0.63	0.08	0.07
TP	2.08	1,32	0.16	0.23	0.26
FT	4.33	1,32	**0.046**	0.33	0.46
Northern Waterthrush					
BGC*TP*FT	1.30	2,48	0.28	0.21	0.24
BGC*TP	0.61	1,48	0.44	0.10	0.11
BGC*FT	0.98	2,48	0.38	0.18	0.18
FT*TP	0.98	2,48	0.38	0.18	0.18
BGC	1.18	1,48	0.28	0.14	0.16
TP	1.81	1,48	0.19	0.17	0.22
FT	0.64	2,48	0.53	0.15	0.14
Orange-crowned Warbler					
BGC*TP*FT	2.16	2,48	0.13	0.27	0.37
BGC*TP	0.06	1,48	0.81	0.03	0.05
BGC*FT	1.77	2,48	0.18	0.24	0.30
FT*TP	0.27	2,48	0.77	0.09	0.08
BGC	4.96	1,48	**0.031**	0.29	0.52
TP	0.30	1,48	0.59	0.07	0.08
FT	7.89	2,48	**0.001**	0.51	0.90

Appendix B (continued)

	3-factor ANOVA statistics				
Species Factors	F	d.f.	P (α)	Effect size (f)	Power ($1-\beta$)
Lincoln's Sparrow					
BGC*TP*FT	1.32	1,32	0.26	0.18	0.17
BGC*TP	0.01	1,32	0.93	0.02	0.05
BGC*FT	7.68	1,32	**0.009**	0.44	0.70
FT*TP	27.63	1,32	**0.000**	0.83	0.99
BGC within Corridor	5.21	1,18	**0.03**	0.51	0.58
BGC within Clearcut	1.84	1,18	0.19	0.30	0.25
FT within MS	19.29	1,18	**0.000**	0.98	0.99
FT within ESSF	3.86	1,18	0.07	0.44	0.46
TP within Corridor	5.21	1,18	**0.03**	0.51	0.58
TP within Clearcut	28.61	1,18	**0.000**	1.20	1.00
FT within Riparian	38.15	1,18	**0.000**	1.38	1.00
FT within Upland	7.43	1,18	**0.01**	0.61	0.73
MacGillivray's Warbler					
BGC*TP*FT	4.62	2,48	**0.015**	0.39	0.67
Within MS-TP*FT	2.21	2,24	0.13	0.38	0.36
Within ESSF-TP*FT	5.23	2,24	**0.013**	0.59	0.73
Within Control-BGC*TP	0.00	1,16	1.00	0.00	0.05
Within Corridor-BGC*TP	15.07	1,16	**0.001**	0.87	0.93
Within Clearcut-BGC*TP	0.37	1,16	0.55	0.14	0.09
Within Riparian-BGC*FT	1.69	2,24	0.21	0.34	0.30
Within Upland-BGC*FT	4.51	2,24	**0.022**	0.55	0.67
One-factor ANOVA	3.23	11,48	**0.002**	0.77	0.98
2.4^{a} 2.4^{a} 2.4^{a} 1.4^{ab} 1.0^{ab} 0.4^{b} 0.4^{b} 0.2^{b} 0^{b} 0^{b} 0^{b} 0^{b}					
Mountain Bluebird					
BGC*TP	1.43	1,16	0.25	0.27	0.19
BGC	0.23	1,16	0.64	0.11	0.07
TP	0.06	1,16	0.81	0.05	0.05

	3-factor ANOVA statistics				
Species Factors	F	d.f.	P (α)	Effect size (f)	Power ($1-\beta$)
Pine Siskin					
BGC*TP*FT	0.65	2,48	0.52	0.15	0.14
BGC*TP	0.41	1,48	0.53	0.08	0.09
BGC*FT	1.65	2,48	0.20	0.23	0.28
FT*TP	2.82	2,48	0.07	0.31	0.47
BGC	10.24	1,48	**0.002**	0.41	0.81
TP	0.00	1,48	1.00	0.00	0.05
FT	1.46	2,48	0.24	0.22	0.26
Red-breasted Nuthatch					
BGC*TP*FT	10.53	2,48	**0.000**	0.59	0.96
Within MS-TP*FT	10.84	2,24	**0.000**	0.85	0.97
Within ESSF-TP*FT	5.81	2,24	**0.009**	0.62	0.76
Within Control-BGC*TP	7.45	1,16	**0.015**	0.61	0.68
Within Corridor-BGC*TP	22.23	1,16	**0.000**	1.05	0.99
Within Clearcut-BGC*TP	0.20	1,16	0.66	0.10	0.07
Within Riparian-BGC*FT	5.68	2,24	**0.010**	0.62	0.76
Within Upland-BGC*FT	27.60	2,24	**0.008**	1.36	1.00
One-factor ANOVA	9.52	11,48	**0.000**	1.32	1.00
7.4^{a} 5.2^{ab} 4.4^{b} 1.0^{c} 1.0^{c} 0.6^{c} 0.6^{c} 0.4^{c} 0.2^{c} 0.2^{c} 0^{c} 0^{c}					
Red Crossbill					
BGC*TP*FT	0.33	2,48	0.72	0.10	0.09
BGC*TP	0.49	1,48	0.49	0.09	0.10
BGC*FT	0.60	2,48	0.55	0.14	0.13
FT*TP	3.17	2,48	0.051	0.33	0.52
BGC	2.44	1,48	0.12	0.20	0.2
TP	11.52	1,48	**0.001**	0.44	0.86
FT	5.45	2,48	**0.007**	0.43	0.76

Appendix B (continued)

3-factor ANOVA statistics

Species Factors	F	d.f.	P (α)	Effect size (f)	Power (1-β)
Ruby-crowned Kinglet					
BGC*TP*FT	10.36	2,48	**0.000**	0.59	0.96
Within MS-TP*FT	3.10	2,24	0.06	0.45	0.49
Within ESSF-TP*FT	7.74	2,24	**0.003**	0.72	0.89
Within Control-BGC*TP	0.57	1,16	0.46	0.17	0.10
Within Corridor-BGC*TP	14.24	1,16	**0.002**	0.84	0.92
Within Clearcut-BGC*TP	0.79	1,16	0.39	0.20	0.13
Within Riparian-BGC*FT	1.75	2,24	0.20	0.34	0.30
Within Upland-BGC*FT	16.05	2,24	**0.000**	1.03	1.00
One-factor ANOVA	8.96	11,48	**0.000**	1.28	1.00
14.0^{a} 8.0^{b} 4.6^{bc} 3.8^{cd} 3.0^{cd} 2.0^{cd} 1.4^{cd} 1.2^{cd} 0.6^{cd} 0.4^{cd} 0.2^{cd} 0^{d}					
Savannah Sparrow					
BGC*TP	0.76	1,16	0.40	0.19	0.12
BGC	0.21	1,16	0.19	0.10	0.07
TP	1.91	1,16	0.65	0.31	0.24
Song sparrow					
BGC*FT	2.41	1,16	0.14	0.35	0.29
BGC	1.61	1,16	0.22	0.28	0.20
FT	4.48	1,16	**0.050**	0.47	0.47
Swainson's Thrush					
BGC*TP*FT	0.44	2,48	0.64	0.12	0.11
BGC*TP	4.11	1,48	**0.048**	0.26	0.44
BGC*FT	0.84	2,48	0.44	0.17	0.17
TP*FT	0.67	2,48	0.51	0.15	0.14
FT	19.05	1,48	**0.000**	0.56	1.00
BGC within Riparian	0.27	1,28	0.61	0.09	0.08
BGC within Upland	3.11	1,28	0.08	0.32	0.39
TP within MS	0.78	1,28	0.38	0.16	0.14
TP within ESSF	1.88	1,28	0.18	0.25	0.26

3-factor ANOVA statistics

Species Factors	F	d.f.	P (α)	Effect size (f)	Power (1-β)
Warbling Vireo					
BGC*TP*FT	3.55	1,32	0.07	0.30	0.40
BGC*TP	0.11	1,32	0.74	0.05	0.06
BGC*FT	1.87	1,32	0.18	0.22	0.24
FT*TP	0.01	1,32	0.91	0.02	0.05
BGC	0.99	1,32	0.33	0.16	0.15
TP	0.07	1,32	0.80	0.04	0.06
FT	4.12	1,32	0.051	0.32	0.44
White-crowned Sparrow					
BGC*TP*FT	0.27	1,32	0.61	0.08	0.07
BGC*TP	0.53	1,32	0.47	0.12	0.10
BGC*FT	0.43	1,32	0.51	0.10	0.08
FT*TP	0.63	1,32	0.43	0.13	0.12
BGC	1.31	1,32	0.26	0.18	0.17
TP	2.02	1,32	0.17	0.22	0.24
FT	21.85	1,32	**0.000**	0.74	0.99
Willow Flycatcher					
BGC*TP	1.37	1,16	0.26	0.26	0.18
BGC	1.97	1,16	0.23	0.31	0.24
TP	4.42	1,16	0.052	0.47	0.47
Wilson's Warbler					
BGC*TP*FT	0.49	2,48	0.62	0.13	0.12
BGC*TP	0.25	1,48	0.62	0.06	0.07
BGC*FT	2.59	2,48	0.09	0.29	0.12
FT*TP	2.86	2,48	0.07	0.31	0.47
BGC	1.49	1,48	0.23	0.16	0.20
TP	9.35	1,48	**0.004**	0.39	0.77
FT	5.22	2,48	**0.009**	0.42	0.74

Appendix B (continued)

3-factor ANOVA statistics					
Species Factors	F	d.f.	P (α)	Effect size (f)	Power (1-β)
Townsend's Warbler					
BGC*TP*FT	0.04	1,32	0.83	0.03	0.05
BGC*TP	16.50	1,32	**0.000**	0.64	0.95
BGC*FT	0.18	1,32	0.68	0.07	0.07
TP*FT	0.55	1,32	0.47	0.12	0.10
FT	0.40	1,32	0.53	0.10	0.09
BGC within Riparian	3.58	1,18	0.07	0.42	0.43
BGC within Upland	25.51	1,18	**0.000**	1.13	1.00
TP within MS	42.44	1,18	**0.000**	1.46	1.00
TP within ESSF	1.14	1,18	0.30	0.24	0.17
Varied Thrush					
BGC*TP*FT	3.22	1,32	0.08	0.28	0.35
BGC*TP	1.21	1,32	0.28	0.17	0.16
BGC*FT	0.03	1,32	0.86	0.03	0.05
FT*TP	2.82	1,32	0.10	0.27	0.33
BGC	5.64	1,32	**0.024**	0.38	0.57
TP	0.08	1,32	0.77	0.04	0.06
FT	0.00	1,32	0.95	0.01	0.05

3-factor ANOVA statistics					
Species Factors	F	d.f.	P (α)	Effect size (f)	Power (1-β)
Winter Wren					
BGC*TP*FT	0.33	2,48	0.72	0.10	0.09
BGC*TP	1.58	1,48	0.22	0.16	0.20
BGC*FT	0.09	2,48	0.91	0.05	0.06
TP*FT	6.68	2,48	**0.003**	0.46	0.82
BGC	0.05	1,48	0.82	0.03	0.05
TP within Control	23.63	1,18	**0.000**	1.09	1.00
TP within Corridor	11.31	1,18	**0.004**	0.75	0.89
TP within Clearcut	0.40	1,18	0.54	0.14	0.09
FT within Riparian	9.60	2,27	**0.001**	0.80	0.97
FT within Upland	1.97	2.27	0.16	0.63	0.84
Yellow-rumped Warbler					
BGC*TP*FT	0.60	2,48	0.55	0.14	0.13
BGC*TP	0.93	1,48	0.34	0.12	0.13
BGC*FT	3.91	2,48	**0.027**	0.36	0.60
FT*TP	3.96	2,48	**0.026**	0.36	0.60
BGC within Control	0.23	1,18	0.64	0.11	0.08
BGC within Corridor	5.04	1,18	**0.038**	0.50	0.56
BGC within Clearcut	1.39	1,18	0.25	0.26	0.20
FT within MS	29.10	2,27	**0.000**	1.39	1.00
FT within ESSF	13.22	2,27	**0.000**	0.94	0.99
TP within Control	7.20	1,18	**0.015**	0.60	0.72
TP within Corridor	0.11	1,18	0.74	0.07	0.06
TP within Clearcut	6.69	1,18	**0.019**	0.58	0.69
FT within Riparian	14.82	2,27	**0.000**	0.99	1.00
FT within Upland	31.93	2,27	**0.000**	1.46	1.00

Appendix C. Mean numbers of bird detections and ANOVA statistics for common bird species and for the total abundance of birds detected in the MS wetland breeding bird investigation. Numbers detected are the sums of detections from six visits to each 2.5-ha plot.

		No. birds detected per plot (N = 3 2.5-ha plots for each cell)				Two-factor ANOVA statistics					
		Topographic position (TP)									
		Wetland		Border							
COMMUNITY MACROPARAMETER	Forest treatment	Mean	S.E.	Mean	S.E.	Effect tested[1]	Mean square	F[2]	P[3] (α)	Effect size[4] (f)	Power (1-β)
Total abundance	Control	73.0	6.8	34.0	5.9	TP*FT	2730.50	4.37	**.037**	0.70	
	Corridor	162.7	15.2	74.7	9.8		$F_{1,4}$	P		$F_{2,6}$	P
	Clearcut	178.7	18.5	54.7	22.3	Control	18.86	**0.012**	Wetland	15.71	**0.004**
						Corridor	23.71	**0.008**	Border	1.97	0.22
						Clearcut	18.28	**0.013**			
Species richness (no. species per plot)	Control	17.0	0.6	13.7	0.3	No separate analysis					
	Corridor	24.3	0.9	17.7	2.6						
	Clearcut	25.0	2.6	17.7	3.2						
SPECIES											
American Robin	Control	0.3	0.3	0.3	0.3	TP*FT	2.72	0.72	0.51	0.28	0.13
	Corridor	1.0	1.0	3.7	1.5	TP	6.72	1.78	0.21	0.31	0.19
	Clearcut	3.3	1.5	4.3	1.5	FT	18.50	4.90	**0.028**	0.74	0.61
Brown-headed Cowbird	Control	1.3	0.7	0.3	0.3	TP*FT	1.17	0.18	0.84	0.14	0.07
	Corridor	4.7	1.8	5.3	1.7	TP	0.00	0.00	1.00	0.00	0.05
	Clearcut	3.0	0.6	3.3	2.4	FT	26.17	4.06	**0.046**	0.67	0.52
Common Snipe	Control	1.0	0.6	0	-	TP*FT	1.50	1.04	0.38	0.34	0.17
	Corridor	2.0	1.2	0	-	TP	18.00	12.46	**0.004**	0.83	0.81
	Clearcut	4.7	0.7	1.7	0.9	FT	12.06	8.35	**0.005**	0.96	0.84
Common Yellowthroat	Control	12.0	5.9	0	-	TP*FT	27.22	0.29	0.75	0.18	0.08
	Corridor	15.3	5.2	1.0	1.0	TP	1088.89	11.48	**0.005**	0.80	0.78
	Clearcut	20.7	11.3	0.3	0.3	FT	30.39	0.32	0.73	0.19	0.08
Dark-eyed Junco	Control	2.7	1.2	5.0	1.5	TP*FT	0.39	0.06	0.94	0.08	0.06
	Corridor	1.7	0.7	5.0	0.6	TP	34.72	5.34	**0.039**	0.54	0.46
	Clearcut	3.0	1.0	5.7	2.7	FT	1.50	0.23	0.80	0.16	0.07
Golden-crowned Kinglet	Control	0	-	5.0	1.5	FT	27.78	5.24	**0.048**	1.08	0.52
	Corridor	0	-	5.3	1.7						
	Clearcut	0	-	0	-						
Lincoln's Sparrow	Control	10.7	1.2	0.3	0.3	TP*FT	16.06	1.81	0.21	0.45	0.26
	Corridor	11.3	3.8	1.0	0.6	TP	320.89	36.10	**0.000**	1.42	1.00
	Clearcut	8.7	0.7	4.0	1.2	FT	1.17	0.13	0.88	0.12	0.06
Mallard	Control	1.7	1.7	0	-	TP*FT	141.72	3.69	0.06	0.64	0.48
	Corridor	18.7	8.5	0	-	TP	249.39	6.50	**0.026**	0.60	0.54
	Clearcut	2.7	2.7	0.7	0.7	FT	131.72	3.43	0.07	0.62	0.46

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[1] TP*FT denotes test of interaction of topographic position (TP) and forest treatment (FT) factors; where TP*FT effect was significant, cell means were compared along rows and columns using one-way ANOVA; where one-way ANOVA was significant and >2 cell means were compared, cell means with like superscripts were not significantly different using Duncan's Multiple Range Test.

[2] Degrees of freedom for F-test: TP Factor (1,12); for FT and TP*FT Factors (2,12).

[3] Significant α denoted in **bold**.

[4] Effect size (f) of Cohen (1988), where 0.4 denotes large effect, 0.25 medium effect, and 0.1 small effect.

Appendix C (continued)

		No. birds detected per plot (N = 3 2.5-ha plots for each cell)				Two-factor ANOVA statistics					
		Topographic position (TP)									
	Forest	Wetland		Border		Effect	Mean		P^3	Effect size[4]	Power
SPECIES	treatment	Mean	S.E.	Mean	S.E.	tested[1]	square	F^2	(α)	(f)	($1-\beta$)
Northern Waterthrush	Control	7.0	1.2	1.0	0.6	TP*FT	4.22	0.51	0.61	0.24	0.10
	Corridor	6.3	2.9	3.7	1.5	TP	80.22	9.69	**0.009**	0.73	0.71
	Clearcut	4.0	2.1	0	-	FT	14.00	1.69	0.23	0.43	0.24
Orange-crowned Warbler	Control	2.7	0.7	0.7	0.3	TP*FT	4.22	0.60	0.56	0.26	0.11
	Corridor	1.3	0.9	0.7	0.7	TP	0.89	0.13	0.73	0.08	0.06
	Clearcut	2.3	1.5	3.7	3.2	FT	6.22	0.89	0.44	0.31	0.14
Pine Siskin	Control	0.7	0.7	1.6	0.9	TP*FT	0.67	0.03	0.97	0.06	0.05
	Corridor	4.7	0.7	7.0	4.6	TP	12.50	0.64	0.44	0.19	0.10
	Clearcut	2.7	1.5	4.3	3.8	FT	32.67	1.67	0.23	0.43	0.24
Ruby-crowned Kinglet	Control	4.0	1.2	2.3	1.5	TP*FT	76.22	29.19	**0.000**	1.80	1.00
	Corridor	0.7	0.3	11.7	0.9						
	Clearcut	1.3	0.9	0.3	0.3						
							$F_{1,4}$	P		$F_{2,8}$	P
						Control	0.81	0.42	Riparian	4.20	0.07
						Corridor	136.13	**0.000**	Upland	36.59	**0.000**
						Clearcut	1.13	0.35			
Red-winged Blackbird	Control	0.3	0.3	0.3	0.3	TP*FT	55.50	0.48	0.63	0.23	0.10
	Corridor	11.3	10.8	0.3	0.3	TP	220.50	1.90	0.19	0.32	0.20
	Clearcut	10.7	10.7	0.7	0.7	FT	58.72	0.51	0.62	0.24	0.10
Song Sparrow	Control	11.7	1.9	0	-	TP*FT	58.72	3.11	0.08	0.59	0.42
	Corridor	20.7	4.7	1.0	1.0	TP	1530.89	81.05	**0.000**	2.12	1.00
	Clearcut	25.7	3.2	1.7	1.2	FT	94.39	5.00	**0.026**	0.75	0.62
Tree Swallow	Control	0.3	0.3	0	-	TP*FT	80.17	0.78	0.48	0.29	0.13
	Corridor	5.0	2.0	0	-	TP	200.00	1.96	0.19	0.33	0.21
	Clearcut	19.0	14.0	4.3	1.8	FT	221.72	2.17	0.16	0.49	0.30
Warbling Vireo	Control	1.0	1.0	0	-	TP*FT	7.06	0.97	0.41	0.33	0.16
	Corridor	5.7	3.0	0.3	0.3	TP	43.56	5.98	**0.031**	0.58	0.51
	Clearcut	4.0	2.1	1.0	0.6	FT	10.50	1.44	0.27	0.40	0.21
Willow Flycatcher	Control	4.7	0.3	0	-	TP*FT	82.89	2.90	0.09	0.57	0.40
	Corridor	16.0	6.0	0	-	TP	773.56	27.04	**0.000**	1.23	0.99
	Clearcut	20.0	4.4	1.3	1.3	FT	108.67	3.80	0.053	0.65	0.50
Wilson's Warbler	Control	6.3	1.9	0	-	TP*FT	0.72	0.03	0.97	0.06	0.05
	Corridor	10.3	5.4	3.7	2.7	TP	168.06	6.41	**0.026**	0.60	0.54
	Clearcut	7.0	3.2	1.7	1.7	FT	23.17	0.88	0.44	0.31	0.14
Yellow Warbler	Control	0.7	0.3	0	-	TP*FT	57.17	2.74	0.10	0.55	0.37
	Corridor	3.3	1.7	0.3	0.3	TP	128.00	6.13	**0.029**	0.56	0.51
	Clearcut	12.3	6.2	0	-	FT	55.06	2.64	0.11	0.54	0.36
Yellow-rumped Warbler	Control	0.7	0.3	4.0	1.0	TP*FT	22.06	6.73	**0.011**	0.86	0.75
	Corridor	1.3	0.9	8.3	1.5						
	Clearcut	2.3	1.3	1.7	0.9						
							$F_{1,4}$	P		$F_{2,8}$	P
						Control	10.00	0.034	Riparian	0.79	0.50
						Corridor	16.96	**0.015**	Upland	8.83	**0.016**
						Clearcut	0.17	0.70			

Appendix D. Mean abundance of all species detected in the MS-ESSF streamside breeding bird investigation. Cn=Control, Cr=Corridor, Cl=Clearcut.

	Mean no. detected per 5-ha plot (N=5 for each mean)											
	MS BGC zone						ESSF BGC zone					
	Streamside			Upland			Streamside			Upland		
SPECIES	Cn	Cr	Cl	Cn	Cr	Cl	Cn	Cr	Cl	Cn	Cr	Cl
American Dipper[1]	0	0	0	0	0	0	0	0	0.2	0	0	0
American Kestrel[3]	0	0	0.6	0	0	0.6	0	0	0.6	0	0	0
*American Robin	0	3.8	5.4	0	2.8	6.2	0.2	1.8	6.4	0.4	0	3.6
Bank Swallow	0	0	0.2	0	0	0	0	0	0	0	0	0
Barn Swallow	0	0	1	0	0	0	0	0	0	0	0	0.4
Belted Kingfisher[1]	0	0	0.6	0	0	0.2	0	0	0	0	0	0
*Boreal Chickadee	0	0	0	0	0	0	0.8	0	0	6	1.2	0
*Brown Creeper	1.6	2	0	0.8	1.8	0	0.4	0.6	0	2.4	1	0
*Brown-headed Cowbird	0	5.4	5.2	0.2	0.6	1	0	0.4	4	0	0.6	1.2
Cassin's Finch[4]	0	0.4	0	0	0	0	0	0	0	0	0	0
Cedar Waxwing[1]	0	0.2	0.2	0	0	1.2	0	0	0	0	0	0
*Chipping Sparrow	0	2	2.8	0	0.6	4.2	0.6	0.8	3.8	2.6	0.4	5.8
Cliff Swallow	0	0	0.8	0	0	0	0	0	0	0	0	0
Common Raven	0	0	0.2	0	0.2	0	0	0	0.2	0	0.4	0
Common Snipe[1]	0	0	0.6	0	0	0	0	0	2.2	0	0	0
Common Yellowthroat[1]	0	0	0.2	0	0	0	0	0	0	0	0	0
*Dark-eyed Junco	3	21.6	22	3.8	11.8	17.2	6	10.8	10	4.8	3.6	18
*Dusky Flycatcher	0	0.6	1	0	0	4.2	0	0.2	0.2	0	0	3
European Starling	0	0	0	0	0	0	0	0	1	0	0	0
*Evening Grosbeak	0.4	1.2	0.2	1.2	3	0	9	1.2	7.2	12.8	0.4	5.2
Fox Sparrow[4]	0	0.2	0	0	0	0	0	0	0.2	0	3	0
*Golden-crowned Kinglet	18.2	15.2	0	8.6	7.6	0	13.6	11.4	0.2	18.6	20	0
*Gray Jay	0.6	2.4	0	2.4	0.8	0	0.2	0.6	1.6	2.4	2.4	2.2
Hairy Woodpecker[4]	0	0	0	0	0.4	0	0	0.2	0.2	0	0.2	0.4
*Hammond's Flycatcher	0.8	3.2	0	0	0	0	0.6	1.2	0	0	0	0
*Hermit Thrush	1.4	3.8	1	0.2	1.8	0.6	4.8	3.6	3.4	3	3.4	1.2
Killdeer[3]	0	0.4	0.2	0	0	1.8	0	0	0.4	0	0	0.4
*Lincoln's Sparrow	0	0	17.4	0	0	4	0	2.2	11.4	0	0	0.6
*MacGillivray's Warbler	0.4	2.4	2.4	0.4	0.2	1	0	0	2.4	0	1.4	0
Mallard[1]	0	0	0.4	0	0	0	0	0	0	0	0	0
Merlin	0	0	0	0	0	0	0	0	0	0	0	0.2
*Mountain Bluebird	0	0	1.8	0	0	0.6	0	0	0.4	0	0	1.2

* For common species, data are also in Appendix A.

For uncommon species:

[1] wetland or stream species; [2] forest species; [3] clearcut species; [4] open-forest species; [5] deciduous tree or shrub species; [6] found in all habitats.

Appendix D (continued)

	Mean no. detected per 5-ha plot (N=5 for each mean)											
	MS BGC zone						ESSF BGC zone					
	Streamside			Upland			Streamside			Upland		
SPECIES	Cn	Cr	Cl	Cn	Cr	Cl	Cn	Cr	Cl	Cn	Cr	Cl
Mountain Chickadee^4	0.6	1.4	0	0.8	0.8	0	0	2.4	0.8	1	1.4	0
Nashville Warbler^5	0	0.8	0	0	0	0.2	0	0	0.2	0	0	0
*Northern Flicker	0	0.8	1	0	0.2	0.6	0	0	1.4	0	0	0.6
Northern Rough-winged Swallow	0	0	0.6	0	0	0	0	0	0.2	0	0	0.4
*Northern Waterthrush	0.6	1	9.8	0.6	1	0	1	1.4	0.4	0.2	0	0
Olive-sided Flycatcher^4	0	0.8	0.4	0	0	0	0	0.6	1.4	0	1	0.4
*Orange-crowned Warbler	0	3	3.4	0.2	1.4	6.4	0	0	2.4	0.4	1.2	1.4
Osprey^1	0	0	0	0	0	0	0	0	0.2	0	0	0
Pileated Woodpecker^2	0	0	0	0	0	0.2	0	0	0	0	0	0
Pine Grosbeak^2	0	0	0	0	0	0	0	0	0	0.4	0.4	0
*Pine Siskin	3.4	12.4	6	9.8	3.8	4.8	14.2	10.6	10.6	18.8	9.8	10.2
*Red Crossbill	1.2	0.8	0	7.2	2.6	0.6	0.2	0	0	3.2	2.4	0
*Red-breasted Nuthatch	0.6	4.4	0	1	0.6	0	0.2	1	0.4	5.2	7.4	0.2
Red-eyed Vireo^5	0	0.2	0	0	0	0	0	0	0	0	0	0
Red-naped Sapsucker^4	0	0	0.8	0	0	1	0	0	0	0	0.2	0
Red-tailed Hawk^4	0	0	0	0	0	0.4	0	0	0.2	0	0	0
*Ruby-crowned Kinglet	0.6	8	1.2	0.2	3	0	1.4	4.6	3.8	2	14	0.4
Ruffed Grouse^6	0	0	0	0	0	0.2	0	0	0	0	0	0
Rufous Hummingbird	0	0.2	0	0	0	0.2	0	0	0	0	0	0.2
*Savannah Sparrow	0	0	1.8	0	0	0.8	0	0	4.4	0	0	0
Solitary Vireo^2	0	0.8	0	0.4	0.2	0	0	0	0	0	0	0
*Song Sparrow	0	0	5.2	0	0	0	0	0.4	1.2	0	0	0
Spotted Sandpiper^1	0	0	0	0	0	0	0	0	0.2	0	0	0
Spruce Grouse^2	0.2	0.2	0	0.2	0	0	0	0	0	0.8	0.6	0
Steller's Jay^2	0	0	0	0	0	0	0	0	0.2	0	0	0
*Swainson's Thrush	2.4	3.2	0.8	1.8	2.6	0	2	3	0.2	3.6	4.8	0
Three-toed Woodpecker^2	0	0.2	0	0	0.4	0	0.2	0.2	0.4	3.2	0.6	0.2
Townsend's Solitaire6	0.2	0.6	0.6	0.6	0.2	0.6	0.2	0.4	0	0.2	0.4	0.4
*Townsend's Warbler	18.6	18.6	0	2.4	1.6	0	16.8	5	0	17.4	13.2	0
Tree Swallow^3	0	0.2	1.6	0	0.2	2.4	0	0	2.4	0	0	0.8
Turkey Vulture	0	0	0.2	0	0	0	0	0	0	0	0	0
Unidentified owl	0	0	0	0	0	0	0	0.2	0	0	0	0
*Varied Thrush	0.6	0.8	0	0	0	0	3.4	0.2	0	1.6	4.4	0

Appendix D (continued)

	Mean no. detected per 5-ha plot (N=5 for each mean)											
	MS BGC zone						ESSF BGC zone					
	Streamside			Upland			Streamside			Upland		
SPECIES	Cn	Cr	Cl	Cn	Cr	Cl	Cn	Cr	Cl	Cn	Cr	Cl
Vesper Sparrow[3]	0	0	0.4	0	0	0.8	0	0	0	0	0	0
Violet-green Swallow	0	0	0.4	0	0	0.2	0	0	0	0	0	0
*Warbling Vireo	0	1.8	4	0	0.2	7.2	0	0.2	3.8	0	2.8	1
Western Meadowlark[3]	0	0	0	0	0	0	0	0	0	0	0	0.2
Western Tanager[2]	0	0	0	0	0	0	0	0.2	0	0	0.2	0
*White-crowned Sparrow	0	0.6	6.8	0	0	5.4	0	1.6	11.2	0	0.4	6.2
*Willow Flycatcher	0	0	6	0	0	0.4	0	0	1.6	0	0	0
*Wilson's Warbler	0.6	6.2	12.8	0.4	0.8	0.6	0.8	14.4	8	1.6	7	1.8
*Winter Wren	6.4	4.8	0.6	0	0	0	5.2	4.8	0	1.6	0.8	0.2
Yellow Warbler[1]	0	0	0.2	0	0	0	0	0	0	0	0	0
*Yellow-rumped Warbler	3.8	16.4	3.2	10.2	17.6	0	5.6	11.4	5	6.8	12	1.6
Total abundance	66.2	153	132	53.4	68.8	75.8	87.4	97.6	116.6	121	123	69.6

Appendix E. Mean abundance of all species detected in the MS wetland breeding bird investigation.

	Mean no. detected per 2.5-ha plot (N=3 for each mean)					
	Wetlands			Borders		
SPECIES	Control	Corridor	Clearcut	Control	Corridor	Clearcut
American Bittern[1]	0	0	0.7	0	0	0
American Crow[1]	0	0	1.0	0	0	0
American Kestrel[3]	0	0	0.3	0	0	0.3
*American Robin	0.3	1.0	3.3	0.3	3.7	4.3
Barn Swallow[3]	0	0	0.7	0	0	2.0
Barrow's Goldeneye[1]	0	1.3	0	0	0	0
Black-capped Chickadee[1]	0	0.3	0	0	0	0
Blue-winged Teal[1]	0	0.7	0	0	0	0
Brown Creeper[2]	0	0	0	0.7	0	0
*Brown-headed Cowbird	1.3	4.7	3.0	0.3	5.3	3.3
Bufflehead[1]	0	0.7	0	0	0	0
Cedar Waxwing[3]	0	0	0.7	0	0.3	0.3
Chipping Sparrow	0	0	0.3	0.3	0.7	0.7
Clark's Nutcracker	0	0	0.7	0	0	0
Common Raven	0	0.3	0	0	0	0.3
*Common Snipe	1.0	2.0	4.7	0	0	1.7
*Common Yellowthroat	12.0	15.3	20.7	0	1.0	0.3
*Dark-eyed Junco	2.7	1.7	3.0	5.0	5.0	5.7
Dusky Flycatcher[3]	0.3	0	0	0	0	0.3
Evening Grosbeak[2]	0	0	0	1.3	0.3	0
*Golden-crowned Kinglet	0	0	0	5.0	5.3	0
Gray Jay[2]	0	0	1.3	1.3	1.0	0
Great Blue Heron[1]	0	0.3	0	0	0	0
Greater Yellowlegs[1]	0	0.7	0.3	0	0	0
Green-winged Teal[1]	0	0	0.7	0	0	0
Hairy Woodpecker	0	0	1.7	0	0	0.3
Hammond's Flycatcher[2]	0.3	0	0	0	0	0
Hermit Thrush[2]	0.3	0	1.0	1.7	2.0	0
Killdeer[3]	0	0	0.3	0	0	1.7
*Lincoln's Sparrow	10.7	11.3	8.7	0.3	1.0	4.0
MacGillivray's Warbler[3]	0.3	1.0	0	0	0	0.7
*Mallard	1.7	18.7	2.7	0	0	0.7
Mountain Bluebird[3]	0	0	3.3	0	0	1.3
Mountain Chickadee[2]	0	0	0	0.7	0	0

*Data for common species are also in Appendix C. For uncommon species: [1]wetland or open-water species; [2]forest or forest-edge species; [3]clearcut species.

Appendix E (continued)

	Mean no. detected per 2.5-ha plot (N=3 for each mean)					
	Wetlands			Borders		
SPECIES	Control	Corridor	Clearcut	Control	Corridor	Clearcut
Northern Flicker[3]	0	0.3	2.7	0	0.3	0
*Northern Waterthrush	7.0	6.3	4.0	1.0	3.7	0
Olive-sided Flycatcher[2]	0	0.3	0	0	0.7	0
*Orange-crowned Warbler	2.7	1.3	2.3	0.7	0.7	3.7
Osprey[1]	0	0	0	0	0	0.3
*Pine Siskin	0.7	4.7	2.7	1.7	7.0	4.3
Red Crossbill[2]	0	0	0	0	0.3	0
Red-breasted Nuthatch[2]	0	0	0.3	0.7	2.0	0
Red-naped Sapsucker	0.3	0	1.3	0	0	0.7
Red-tailed Hawk	0	0.3	0	0	0	0
*Red-winged Blackbird	0.3	11.3	10.7	0.3	0.3	0.7
Ring-necked Duck[1]	0	5.0	0	0	0	0
*Ruby-crowned Kinglet	4.0	0.7	1.3	2.3	11.7	0.3
Rusty Blackbird[1]	0	0.7	0.3	0	0	0
Savannah Sparrow[3]	0	1.0	0.7	0	0	1.7
Solitary Sandpiper[1]	1.0	2.7	0	0	0	0
Solitary Vireo[2]	0	0	0	0.7	0	0
*Song Sparrow	11.7	20.7	25.7	0	1.0	1.7
Sora[1]	0	2.0	1.7	0	0	0
Spotted Sandpiper[1]	0	0.3	0	0	0	0
Spruce Grouse[2]	0	0.3	0	0.3	0	0
Swainson's Thrush	0.7	0.7	0.3	1.0	1.7	0
Townsend's Solitaire[3]	0	0	0	0	0	1.3
Townsend's Warbler[2]	0	0	0	2.7	2.0	0
*Tree Swallow	0.3	5.0	19.0	0	0	4.3
Varied Thrush[2]	0	0.3	0	1.7	3.0	0
Vesper Sparrow[3]	0	0	0	0	0	0.3
Violet-green Swallow[1]	0	0	0.3	0	0	0
*Warbling Vireo	1.0	5.7	4.0	0	0.3	1.0
Western Wood-Pewee	0	0	0.3	0	0	0
White-crowned Sparrow[3]	0	1.7	0.3	0	0.7	1.7
*Willow Flycatcher	4.7	16.0	20	0	0	1.3
*Wilson's Warbler	6.3	10.3	7.0	0	3.7	1.7
Winter Wren[2]	0	0.3	0	0	1.3	0
*Yellow Warbler	0.7	3.3	12.3	0	0.3	0
*Yellow-rumped Warbler	0.7	1.3	2.3	4.0	8.3	1.7
Total abundance	73.0	162.7	178.7	34.0	74.7	54.7

Appendix F. **Species list of all breeding birds encountered during the breeding season in the MS and ESSF of the Okanagan Highlands, grouped according to dependence on riparian areas and the potential to be affected by timber harvesting.**

Species reliant on riparian areas

Riparian species greatly affected by timber harvest

- Golden-crowned Kinglet
- Hammond's Flycatcher
- Townsend's Warbler
- Varied Thrush
- White-winged Crossbill
- Winter Wren

Riparian species somewhat affected by timber harvest (require trees or healthy streams, but not necessarily forests)

- American Dipper
- American Crow
- Barrow's Goldeneye
- Belted Kingfisher
- Black-capped Chickadee
- Bufflehead
- Great Blue Heron
- Northern Waterthrush
- Osprey

Riparian species not negatively affected by timber harvest

- American Bittern
- Blue-winged Teal
- Brown-headed Cowbird
- Cedar Waxwing
- Common Loon
- Common Snipe
- Common Yellowthroat
- Greater Yellowlegs
- Green-winged Teal
- Lincoln's Sparrow
- MacGillivray's Warbler
- Mallard
- Red-winged Blackbird
- Ring-necked Duck
- Rusty Blackbird
- Solitary Sandpiper
- Song Sparrow
- Sora
- Spotted Sandpiper
- Willow Flycatcher
- Wilson's Warbler
- Yellow Warbler

Species not reliant on riparian areas

Species greatly affected by timber harvest

- Black-backed Woodpecker
- Boreal Chickadee
- Brown Creeper
- Cooper's Hawk
- Gray Jay
- Hairy Woodpecker
- Merlin
- Pileated Woodpecker
- Pine Grosbeak
- Red Crossbill
- Red-breasted Nuthatch
- Solitary Vireo
- Swainson's Thrush
- Three-toed Woodpecker
- Vaux's Swift
- Western Tanager
- Western Wood-Pewee
- White-breasted Nuthatch

Species somewhat affected by timber harvest (require trees, but not necessarily forests)

- American Kestrel
- Cassin's Finch
- Clark's Nutcracker
- Evening Grosbeak
- Fox Sparrow
- Great Gray Owl
- Hermit Thrush
- Mountain Chickadee
- Mountain Bluebird
- Northern Flicker
- Olive-sided Flycatcher
- Pine Siskin
- Red-eyed Vireo
- Red-naped Sapsucker
- Red-tailed Hawk
- Ruby-crowned Kinglet
- Spruce Grouse
- Steller's Jay
- Tree Swallow
- Yellow-rumped Warbler

Species not negatively affected by timber harvest

- American Robin
- Bank Swallow
- Barn Swallow
- Chipping Sparrow
- Cliff Swallow
- Common Raven
- Common Nighthawk
- Dark-eyed Junco
- Dusky Flycatcher
- European Starling
- Golden Eagle
- Killdeer
- Nashville Warbler
- Northern Rough-winged Swallow
- Orange-crowned Warbler
- Rock Wren
- Ruffed Grouse
- Rufous Hummingbird
- Savannah Sparrow
- Townsend's Solitaire
- Turkey Vulture
- Vesper Sparrow
- Violet-green Swallow
- Warbling Vireo
- Western Meadowlark
- White-crowned Sparrow

Appendix G. Species list of breeding birds encountered during forest plot sampling in the MS and ESSF of the Okanagan Highlands, grouped according to required forested conditions and abundance.

Forest Species (generally requiring closed forests)

Width-Sensitive Forest Species
- Boreal Chickadee
- Brown Creeper
- Gray Jay
- Swainson's Thrush
- Townsend's Warbler
- Winter Wren

Common Forest Species
- Evening Grosbeak
- Golden-crowned Kinglet
- Red-breasted Nuthatch
- Red Crossbill
- Varied Thrush

Uncommon Forest Species
- Black-backed Woodpecker
- Hairy Woodpecker
- Nashville Warbler
- Pine Grosbeak
- Red-eyed Vireo
- Solitary Vireo
- Spruce Grouse
- Three-toed Woodpecker
- White-breasted Nuthatch
- Western Tanager
- Western Wood-Pewee

Species able to use open forest or very young regenerating forests

Open Forest Species
- Cassin's Finch
- Fox Sparrow
- Hammond's Flycatcher
- Hermit Thrush
- Mountain Chickadee
- Olive-sided Flycatcher
- Red-naped Sapsucker
- Red-tailed Hawk
- Ruby-crowned Kinglet
- Wilson's Warbler
- Yellow-rumped Warbler

Appendix H. Full two-factor ANOVA statistics for comparisons of mean small mammal abundance and species richness.

		No. small mammals caught /100 TN (N = 5 transects for each mean)				Two-factor ANOVA statistics					
		Topographic position (TP)									
		Riparian		Upland							
Species	Forest treatment	Mean	S.E.	Mean	S.E.	Effect tested[1]	Mean square	F[2]	P[3] (α)	Effect size[4] (f)	Power (1-β)
Common Shrew	Control	0.8	0.6	1.8	1.3	TP*FT	5.20	2.04	0.15	0.37	0.35
	Corridor	1.8	1.0	0	-	TP	0.30	0.12	0.73	0.06	0.06
	Clearcut	0	-	0.2	0.2	FT	3.73	1.46	0.25	0.31	0.25
Deer Mouse	Control	0	-	0	-	TP*FT	2.53	1.50	0.24	0.32	0.27
	Corridor	1.0	0.4	2.2	0.5	TP	8.53	5.07	**0.04**	0.41	0.52
	Clearcut	0.2	0.2	2.2	1.2	FT	6.93	4.12	**0.03**	0.52	0.62
Dusky Shrew	Control	0.2	0.2	0	-	TP*FT	0.03	0.50	0.61	0.18	0.11
	Corridor	0.2	0.2	0	-	TP	0.13	2.00	0.17	0.37	0.37
	Clearcut	0	-	0	-	FT	0.03	0.50	0.61	0.18	0.11
Heather Vole	Control	1.2	1.2	0.2	0.2	TP*FT	1.20	1.18	0.32	0.28	0.21
	Corridor	1.0	0.4	0	-	TP	2.70	2.66	0.12	0.30	0.31
	Clearcut	0	-	0.2	0.2	FT	0.93	0.92	0.41	0.25	0.18
Long-tailed Vole	Control	2.8	1.5	0	-	TP*FT	3.33	1.47	0.25	0.31	0.25
	Corridor	0.8	0.6	0	-	TP	16.13	7.12	**0.01**	0.49	0.67
	Clearcut	0.8	0.2	0	-	FT	3.33	1.47	0.25	0.31	0.25
Meadow Vole	Control	0.2	0.2	0	-	TP*FT	30.90	3.19	0.06	0.46	0.51
	Corridor	1.4	1.2	0	-	TP	58.80	6.07	**0.02**	0.35	0.60
	Clearcut	7.4	3.2	0.6	0.4	FT	44.10	4.55	**0.02**	0.55	0.67
Red-backed Vole	Control	10.0	1.9	10.6	1.9	TP*FT	25.83	2.22	0.13	0.39	0.38
	Corridor	7.2	2.4	1.8	0.9	TP	22.53	1.94	0.18	0.25	0.23
	Clearcut	0.6	0.4	0.2	0.2	FT	494.87	21.30	**0.000**	1.19	1.00
Vagrant Shrew	Control	2.6	1.2	0.6	0.2	TP*FT	8.03	1.67	0.21	0.33	0.28
	Corridor	5.2	1.5	0	-	TP	73.63	15.29	**0.000**	0.71	0.93
	Clearcut	3.6	1.3	1.4	0.7	FT	3.03	0.63	0.54	0.20	0.13
Water Shrew	Control	0	-	0	-	TP*FT	1.20	1.53	0.24	0.32	0.27
	Corridor	1.2	1.0	0	-	TP	1.20	1.53	0.23	0.23	0.20
	Clearcut	0	-	0	-	FT	1.20	1.53	0.24	0.32	0.27
Yellow-pine Chipmunk	Control	0	-	0	-	TP*FT	0.53	1.00	0.38	0.26	0.19
	Corridor	0.4	0.2	0.4	0.2	TP	0.53	1.00	0.33	0.18	0.14
	Clearcut	0.6	0.4	1.4	0.6	FT	2.53	4.75	**0.02**	0.56	0.69
Total abundance	Control	17.8	2.1	13.2^a	2.6	TP*FT	86.93	3.31	**0.05**	0.47	0.53
	Corridor	20.2	2.1	4.4^b	1.0	TP within	$F_{1,8}$	P	FT within	$F_{2,12}$	P
	Clearcut	13.2	3.3	6.2^b	0.6	Control	1.93	0.20	Riparian	1.92	0.19
						Corridor	45.39	**0.000**	Upland	5.52	**0.020**
						Clearcut	3.25	0.11			
Species richness (S)	Control	3.6	0.4	2.4	0.5	TP*FT	4.90	5.76	**0.009**	0.62	0.78
	Corridor	5.4	0.5	2.0	0.0	TP within	$F_{1,8}$	P	FT within	$F_{2,12}$	P
	Clearcut	4.0	0.4	3.2	0.4	Control	3.43	0.10	Riparian	4.32	0.04
						Corridor	44.46	**0.000**	Upland	2.80	0.10
						Clearcut	1.88	0.21			

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[1] TP*FT denotes test of interaction of Topographic Position (TP) and Forest Treatment (FT) factors; where TP*FT effect was significant, cell means were compared along rows and columns using one-way ANOVA; where one-way ANOVA significant and >2 cell means were compared, cell means with like superscripts not significantly different using Duncan's Multiple Range Test.

[2] Degrees of Freedom for F-test: TP Factor (1,24); for FT and TP*FT Factors (2,24).

[3] Significant α denoted in **bold**.

[4] Effect Size (f) of Cohen (1988) where 0.4 denotes large effect, 0.25 medium effect, and 0.1 small effect.

Appendix I. **List of common and scientific names for bird species mentioned.**

Common Name	Scientific Name
American Bittern	*Botaurus lentiginosus* (Rackett)
American Crow	*Corvus brachyrhynchos* Brehm
American Dipper	*Cinclus mexicanus* Swainson
American Kestrel	*Falco sparverius* Linnaeus
American Robin	*Turdus migratorius* Linnaeus
Bank Swallow	*Riparia riparia* (Linnaeus)
Barrow's Goldeneye	*Bucephala islandica* (Gmelin)
Belted Kingfisher	*Ceryle alcyon* (Linnaeus)
Black-backed Woodpecker	*Picoides arcticus* (Swainson)
Black-capped Chickadee	*Poecile atricapillus* Linnaeus
Blue-winged Teal	*Anas discors* Linnaeus
Boreal Chickadee	*Poecile hudsonicus* (Forster)
Brown Creeper	*Certhia americana* Bonaparte
Brown-headed Cowbird	*Molothrus ater* (Boddaert)
Bufflehead	*Bucephala albeola* (Linnaeus)
Cassin's Finch	*Carpodacus cassinii* Baird
Cedar Waxwing	*Bombycilla cedrorum* Vieillot
Chipping Sparrow	*Spizella passerina* (Bechstein)
Clark's Nutcracker	*Nucifraga columbiana* (Wilson)
Cliff Swallow	*Petrochelidon pyrrhonota* Vieillot
Common Loon	*Gavia immer* (Brünnich)
Common Nighthawk	*Chordeiles minor* (Forster)
Common Raven	*Corvus corax* Linnaeus
Common Snipe	*Gallinago gallinago* (Linnaeus)
Common Yellowthroat	*Geothlypis trichas* (Linnaeus)
Cooper's Hawk	*Accipiter cooperii* (Bonaparte)
Dark-eyed Junco	*Junco hyemalis* (Linnaeus)
Dusky Flycatcher	*Empidonax oberholseri* Phillips
European Starling	*Sturnus vulgaris* Linnaeus
Evening Grosbeak	*Coccothraustes vespertinus* (Cooper)
Fox Sparrow	*Passerella iliaca* (Merrem)
Golden Eagle	*Aquila chrysaetos* (Linnaeus)
Golden-crowned Kinglet	*Regulus satrapa* Lichtenstein
Gray Jay	*Perisoreus canadensis* (Linnaeus)
Great Blue Heron	*Ardea herodias* Linnaeus
Great Gray Owl	*Strix nebulosa* Forster
Greater Yellowlegs	*Tringa melanoleuca* (Gmelin)
Green-winged Teal	*Anas crecca* Linnaeus
Hairy Woodpecker	*Picoides villosus* (Linnaeus)
Hammond's Flycatcher	*Empidonax hammondii* (Xántus de Vesey)
Hermit Thrush	*Catharus guttatus* (Pallas)
Killdeer	*Charadrius vociferus* Linnaeus
Lincoln's Sparrow	*Melospiza lincolnii* (Audubon)
MacGillivray's Warbler	*Oporornis tolmiei* (Townsend)
Mallard	*Anas platyrhynchos* Linnaeus
Merlin	*Falco columbarius* Linnaeus
Mountain Bluebird	*Sialia currucoides* (Bechstein)
Mountain Chickadee	*Poecile gambeli* (Ridgway)
Nashville Warbler	*Vermivora ruficapilla* (Wilson)
Northern Flicker	*Colaptes auratus* (Linnaeus)
Northern Rough-winged Swallow	*Stelgidopteryx serripennis* (Audubon)
Northern Waterthrush	*Seiurus noveboracensis* (Gmelin)
Olive-sided Flycatcher	*Contopus cooperi* (Nuttall)
Orange-crowned Warbler	*Vermivora celata* (Say)
Osprey	*Pandion haliaetus* (Linnaeus)
Pileated Woodpecker	*Dryocopus pileatus* (Linnaeus)
Pine Grosbeak	*Pinicola enucleator* (Linnaeus)
Pine Siskin	*Carduelis pinus* (Wilson)
Red Crossbill	*Loxia curvirostra* Linnaeus
Red-breasted Nuthatch	*Sitta canadensis* Linnaeus
Red-eyed Vireo	*Vireo olivaceus* (Linnaeus)
Red-naped Sapsucker	*Sphyrapicus nuchalis* Baird

Appendix I (continued)

Common Name	Scientific Name
Red-tailed Hawk	*Buteo jamaicensis* (Gmelin)
Red-winged Blackbird	*Agelaius phoeniceus* (Linnaeus)
Ring-necked Duck	*Aythya collaris* (Donovan)
Rock Wren	*Salpinctes obsoletus* (Say)
Ruby-crowned Kinglet	*Regulus calendula* (Linnaeus)
Ruffed Grouse	*Bonasa umbellus* (Linnaeus)
Rufous Hummingbird	*Selasphorus rufus* (Gmelin)
Rusty Blackbird	*Euphagus carolinus* (Müller)
Savannah Sparrow	*Passerculus sandwichensis* (Gmelin)
Solitary Sandpiper	*Tringa solitaria* Wilson
Solitary Vireo [Blue-headed Vireo]	*Vireo solitarius* (Wilson)
Song Sparrow	*Melospiza melodia* (Wilson)
Sora	*Porzana carolina* (Linnaeus)
Spotted Sandpiper	*Actitis macularia* (Linnaeus)
Spruce Grouse	*Falcipennis canadensis* (Linnaeus)
Steller's Jay	*Cyanocitta stelleri* (Gmelin)
Swainson's Thrush	*Catharus ustulatus* (Nuttall)
Three-toed Woodpecker	*Picoides tridactylus* (Linnaeus)
Townsend's Solitaire	*Myadestes townsendi* (Audubon)
Townsend's Warbler	*Dendroica townsendi* (Townsend)
Tree Swallow	*Tachycineta bicolor* (Vieillot)
Turkey Vulture	*Cathartes aura* (Linnaeus)
Varied Thrush	*Ixoreus naevius* (Gmelin)
Vaux's Swift	*Chaetura vauxi* (Townsend)
Vesper Sparrow	*Pooecetes gramineus* (Gmelin)
Violet-green Swallow	*Tachycineta thalassina* (Swainson)
Warbling Vireo	*Vireo gilvus* (Vieillot)
Western Meadowlark	*Sturnella neglecta* Audubon
Western Tanager	*Piranga ludoviciana* (Wilson)
Western Wood-Pewee	*Contopus sordidulus* Sclater
White-breasted Nuthatch	*Sitta carolinensis* Latham
White-crowned Sparrow	*Zonotrichia leucophrys* (Forster)
White-winged Crossbill	*Loxia leucoptera* Gmelin
Willow Flycatcher	*Empidonax traillii* (Audubon)
Wilson's Warbler	*Wilsonia pusilla* (Wilson)
Winter Wren	*Troglodytes troglodytes* (Linnaeus)
Yellow Warbler	*Dendroica petechia* (Linnaeus)
Yellow-rumped Warbler	*Dendroica coronata* (Linnaeus)

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